

BOSQUINES D' *ARBUTUS UNEDO* L.: INCREMENT DE LA
SEVA PRESÈNCIA PEL RÈGIM DE FOCS I EFECTES DE LA
GESTIÓ SILVÍCOLA POSTINCENDI SOBRE EL SEU
CREIXEMENT, REPRODUCCIÓ I FAUNA ASSOCIADA

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CERTIFIQUEN:

Que aquest treball, realitzat a la Unitat d'Ecologia del BAVE i el CREA F, i titulat "Bosquines d'*Arbutus unedo* L.: increment de la seva presència pel règim de focs i efectes de la gestió silvícola postincendi sobre el seu creixement, reproducció i fauna associada", que presenta Lídia Quevedo i Dalmau per a l'obtenció del títol de Doctora, ha estat realitzat sota la seva direcció.

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RESUM

En les darreres dècades, el règim d'incendis forestals a la Conca Mediterrània ha canviat, produint-se un augment del número de focs i un increment de l'extensió de la superfície cremada (80-90%) per part d'uns pocs incendis (<8%). Una elevada freqüència de focs i unes condicions climàtiques postincendi extremes poden donar lloc a què l'espècie arbòria dominant de la coberta forestal anterior a l'incendi tingui problemes per recuperar-se després del foc, produint-se un canvi de coberta forestal. Ja sigui en aquest context o quan un foc afecta una coberta forestal dominada per rebrotadores, després del foc es poden desenvolupar bosquines d'espècies rebrotadores, amb una elevada densitat d'individus per hectàrea i de rebrots per individu. Aquestes bosquines d'alta densitat presenten un estancament en el seu creixement i un elevat risc d'incendi per la continuïtat de combustible. L'augment aparent de superfície forestal coberta per bosquines d'arboç (*Arbutus unedo* L.) a Catalunya, així com el fet que la informació de com gestionar aquesta espècie en escenaris postincendi sigui molt escassa, fa rellevant el contingut d'aquesta tesi.

La present tesi té els següents objectius: 1) analitzar quin és el paper dels incendis forestals en la formació de les bosquines dominades per arboç a Catalunya i si hi ha hagut un augment de la superfície ocupada per aquesta coberta; 2) analitzar quins efectes té l'aplicació de tractaments silvícoles (selecció de rebrots d'arboç, acompanyada o no de la desbrossada del sotabosc), sobre el creixement i la reproducció d'una població d'arboç, durant tres anys de seguiment, i els impactes sobre la diversitat de la fauna associada. Aquests tractaments s'han aplicat en una zona cremada a Catalunya en l'hivern de finals del 2006 i inici del 2007.

En aquesta tesi s'ha pogut demostrar com s'ha produït un augment de la superfície coberta per bosquines d'arboç a Catalunya, ja que mitjançant anàlisis SIG mostrem com al 1996 aquestes bosquines havien augmentat clarament la seva superfície respecte la que tenien al 1979. A més a més hem constatat que han aparegut bàsicament en pinedes de pi blanc cremades recurrentment.

Pel que fa a la resposta de les bosquines d'arboç als tractaments silvícoles, hem demostrat que la selecció de rebrots promou un major creixement dels rebrots reservats, de manera que creixen un 48% en longitud i un 60% més en diàmetre respecte els rebrots dels individus control. Aquest efecte ja es dona a curt termini i es manté els tres anys de seguiment. El vigor de la rebrotada induïda pel tractament de selecció no augmenta pel fet de desbrossar la vegetació al voltant de l'arbre. Tampoc s'ha trobat una relació negativa entre el vigor de la rebrotada induïda i el

creixement dels rebrots reservats, com es dona en d'altres espècies. Per altra banda, els arboços de les parcel·les amb selecció de rebrots i desbrossada del sotabosc, tot i no haver produït més fruits madurs, sí que han estat de major qualitat (major pes sec i amb més llavors desenvolupades per fruit). La quantitat de fruits per hectàrea no ha variat entre tractaments, de manera que es garanteix la font d'aliment per a la fauna de la comunitat i el potencial d'establiment de plàntules d'arboç. Finalment, els tractaments no han alterat l'estructura, ni la composició ni els gremis alimentaris de la comunitat de formigues que habita a les bosquines d'arboç, les quals s'han emprat com a bioindicadores per avaluar un possible impacte d'aquests tractaments sobre la fauna local.

Hem demostrat que els tractaments descrits milloren l'estructura dels arboçars i estimulen el seu creixement, potenciant per tant l'aprofitament dels seus recursos forestals. A més, la major qualitat dels seus fruits podria donar lloc a un increment en l'establiment de plàntules. Per tant, les cada vegada més presents bosquines d'arboç en el nostre territori poden ser gestionades satisfactòriament per obtenir-ne un aprofitament i una correcta prevenció d'incendis forestals, sense comprometre la seva regeneració ni alterar la diversitat de la fauna associada.

RESUMEN

En las últimas décadas, el régimen de incendios forestales en la Cuenca Mediterránea ha cambiado, produciéndose un aumento del número de fuegos y un incremento de la extensión de la superficie quemada (80-90%) por parte de unos pocos incendios (<8%). Una elevada frecuencia de fuegos y unas condiciones climáticas postincendio extremas pueden dar lugar a que la especie arbórea dominante de la cubierta forestal anterior al incendio tenga problemas para recuperarse después del fuego, produciéndose un cambio de cubierta forestal. Ya sea en este contexto o cuando un fuego afecta una cubierta forestal dominada por rebrotadoras, después del fuego se puede desarrollar monte bajo de especies rebrotadoras, con una elevada densidad de individuos por hectárea y de rebrotes por individuo. Este monte bajo de alta densidad presenta un estancamiento en su crecimiento y un elevado riesgo de incendio por la continuidad de combustible. El aumento aparente de superficie forestal cubierta por monte bajo de madroño (*Arbutus unedo* L.) en Catalunya, así como el hecho que la información de cómo gestionar esta especie en escenarios postincendio sea muy escasa, hace relevante el contenido de esta tesis.

Esta tesis tiene los siguientes objetivos: 1) analizar cuál es el papel de los incendios forestales en la formación del monte bajo de madroño en Catalunya y si ha habido un aumento de la superficie ocupada por esta cubierta; 2) analizar qué efectos tiene la aplicación de tratamientos silvícolas (resalveo del madroño, acompañado o no del desbroce del sotobosque), sobre el crecimiento y la reproducción de una población de madroño, y los impactos sobre la diversidad de la fauna asociada. Estos tratamientos se han aplicado en una zona quemada en Catalunya en el invierno de finales del 2006 y principios del 2007.

En esta tesis se ha podido demostrar como se ha producido un aumento de la superficie cubierta por monte bajo de madroño en Catalunya, ya que mediante análisis SIG hemos constatado un claro aumento de la superficie ocupada por monte bajo de madroño en 1996 respecto la que ocupaba en 1979. Además mostramos como ha aparecido básicamente en pinares de pino carrasco quemadas recurrentemente.

En lo que se refiere a la respuesta del monte bajo de madroño a los tratamientos silvícolas, hemos comprobado que el resalveo promueve un mayor crecimiento de los rebrotes reservados, de forma que crecen un 48% en longitud y un 60% más en diámetro respecto los rebrotes de los individuos control. Este efecto ya se da a corto plazo y se mantiene durante los tres años de seguimiento. El vigor del rebrote

inducido por el tratamiento de resalveo no aumenta por el hecho de desbrozar la vegetación alrededor del árbol. Tampoco se ha encontrado una relación negativa entre el vigor del rebrote inducido y el crecimiento de los rebrotes reservados, tal como se da en otras especies. Por otro lado, los madroños de las parcelas con resalveo y desbroce del sotobosque, aunque no han producido más frutos maduros, sí que han sido de mayor calidad (mayor peso seco y con más semillas desarrolladas por fruto). La cantidad de frutos por hectárea no ha variado entre tratamientos, de forma que podemos afirmar que se garantiza la fuente de alimento para la fauna de la comunidad y el potencial de establecimiento de plántulas de madroño. Finalmente, los tratamientos no han alterado la estructura, ni la composición ni los gremios alimentarios de la comunidad de hormigas que habita en el monte bajo de madroño, las cuales se han utilizado como bioindicadoras para evaluar un posible impacto de estos tratamientos sobre la fauna local.

Por tanto, hemos demostrado que los tratamientos descritos mejoran la estructura del monte bajo de madroño y estimulan su crecimiento, potenciando el aprovechamiento de sus recursos forestales. Además, la mayor calidad de sus frutos podría dar lugar a un incremento en el establecimiento de plántulas. Por tanto, el cada vez más presente monte bajo de madroño en nuestro territorio puede ser gestionado satisfactoriamente para obtener un aprovechamiento y una correcta prevención de incendios forestales, sin comprometer su regeneración ni alterar la diversidad de la fauna asociada.

SUMMARY

In recent decades, the fire regime in the Mediterranean Basin has changed, with an increase in fire recurrence and a greater proportion of area burned in large intense fires (80-90% burned area in <8% fires). High fire recurrence and extreme post-fire climatic conditions may hinder recovery of pre-fire dominant tree species, thus producing a change in forest cover. In this context, especially when fire affects areas dominated by resprouting species, post-fire coppices may display high tree densities and a large number of resprouts per tree. These high-density coppices show growth stagnation and entail great fire propagation risk due to fuel continuity. This dissertation addressed the apparent increase of forest cover by the resprouter strawberry-tree (*Arbutus unedo* L.) in Catalonia. The study is relevant because there is little information available on how to manage this species in a post-fire scenario.

The objectives of this thesis are: 1) to analyze the role of fire in the occurrence of coppices dominated by the resprouter strawberry-tree and to establish whether there has been an increase in the area occupied by these coppices; 2) to analyze the effects of forestry management practices (selective thinning of resprouts and selective thinning of resprouts plus mechanical understory clearing) on growth and reproduction of a strawberry-tree population and on animal diversity during three years. These treatments were applied to a burned area in Catalonia in late 2006 to early 2007.

GIS analysis shows a clear increase in the surface occupied by strawberry-tree coppices in Catalonia from 1979 to 1996. These increases have mainly occurred in repeatedly burned Aleppo pine forests.

Regarding the effect of selective thinning applied to strawberry tree coppices, we show that these treatments promoted higher resprout growth, so that retained resprouts of treated strawberry trees grew 48% longer and 60% thicker than those of control trees. These effects occur in the short and intermediate terms. Induced resprouting by thinning was unaffected by the type of understory clearing treatment applied. In contrast to other species, we failed to find a negative relationship between the degree of induced resprouting and retained resprout growth. On the other hand, production of mature fruits did not vary among treatments, strawberry-trees from plots with selective thinning plus understory clearing produced fruits of higher quality (grater dry weight and seed-set per fruit). Mature fruit production per hectare was not

affected by treatments. Therefore, management practices may enhance the natural regeneration of populations of this species, as well as the availability of food for local frugivorous fauna. Finally, treatments had no impact on structure, composition and dietary guilds of the ant communities living in strawberry-tree coppices. These communities were used as bioindicators to assess the impact of these forestry treatments on local fauna.

In sum, we demonstrate that forestry practices improve the structure of strawberry-tree coppices and stimulate their growth, promoting the use of fruit resources. In addition, the increase in fruit quality could entail an increase in seedling establishment. Therefore, the increasing strawberry-tree coppices present in our territory may be satisfactorily managed to obtain good use and correct forest fire prevention, without affecting neither their regeneration nor the diversity of the associated fauna.

INTRODUCCIÓ GENERAL



INTRODUCCIÓ GENERAL

La gestió forestal en els ecosistemes mediterranis

Tradicionalment els boscos mediterranis han estat explotats intensament per a l'obtenció de fusta i llenyes. Aquests productes forestals foren utilitzats tant per a l'elaboració de mobles, vaixells, eines, etc., així com per a la producció de calor a les llars i forns de pa. Les principals espècies arbòries mediterrànies emprades per aquestes finalitats han estat *Pinus halepensis*, *P. pinea*, *P. sylvestris*, *Quercus ilex*, *Q. x cerrioides*, entre d'altres (Diputació de Barcelona et al. 2009, Obra Social "La Caixa" et al. 2010). A la vegada, aquests aprofitaments forestals també milloren el valor paisatgístic de les comunitats forestals i en faciliten l'ús de pastures (Rigueiro-Rodríguez et al. 2009, López-Díaz et al. 2013).

Tot i així, la terciarització de l'economia durant la segona meitat del segle XX (Blondel i Aronson 1999) produí l'abandonament de camps agrícoles, que juntament amb una reducció dels aprofitaments de productes forestals, provocà que la superfície forestal de Catalunya augmentés en un 5,4% entre els anys 1993 i 2009, essent de 2.058.133 hectàrees a l'any 2009 (CREAF i Generalitat de Catalunya 1993, 2009). Aquestes grans masses forestals que configuren els paisatges mediterranis actuals es caracteritzen per una elevada continuïtat horitzontal i vertical de combustible, fet que augmenta el risc d'incendi. A més, altres elements de risc d'incendi actuals al bosc són l'augment de la freqüentació humana a les zones forestals, ja sigui pels grans processos d'urbanització que s'han produït, com pels creixents usos recreatius dels boscos; i, per altra banda, el canvi climàtic està provocant una disminució de les precipitacions, més evapotranspiració i estius més llargs en els ecosistemes mediterranis, així com més esdeveniments extraordinaris de sequera (Kattenberg et al. 1996, Piñol et al. 1998, Bates et al. 2008). Per tant, hi ha la necessitat d'aplicar tractaments de gestió forestal en gran part dels boscos mediterranis per reduir aquesta situació de risc d'incendi elevada. Alguns d'aquests tractaments es porten realitzant tan sols des de fa poques dècades i sovint de manera experimental sobre superfícies relativament petites, i per espècies d'àmplia distribució (Moreira et al. 2011), pel que encara no hi ha un bon coneixement de la seva efectivitat a gran escala i considerant altres espècies més minoritàries.

El foc en els ecosistemes mediterranis

Aquesta situació de risc d'incendi tan gran en els ecosistemes mediterranis ha contribuït en que, en les darreres dècades, el règim d'incendis hagi canviat, de forma que s'ha produït un augment anual del número de focs, un increment de la superfície cremada per part dels anomenats Grans incendis forestals (GIF, incendis que afecten més de 1.000 hectàrees), de manera que només un 8% dels focs són els responsables del 80-90% de la superfície anual cremada; i un augment de l'ocurrència d'incendis en àrees més submediterrànies que fins ara no havien estat afectades per incendis (Piñol et al. 1998, Pausas 2004, Lloret et al. 2009, Moreira et al. 2011).

El foc ha estat sempre una de les principals perturbacions en els ecosistemes mediterranis i ha modificat les característiques del paisatge en funció de la seva recurrència, intensitat i superfície afectada (Tsitsoni 1997, Pausas et al. 2003, Broncano i Retana 2004, Eugenio i Lloret 2004, Paula i Ojeda 2006). Per exemple, poden provocar la desaparició total o parcial de la cobertura vegetal i la pèrdua de les característiques estructurals del bosc (Piussi 1992, Hutto i Gallo 2006, Herrando et al. 2009, Moreira et al. 2011). També poden donar lloc a canvis en la composició de les espècies vegetals, i quan aquests canvis es donen en les espècies dominants del vol del bosc, repercuteixen a escala de paisatge (Trabaud 1994, Rodrigo et al. 2004) i alterar el seu règim de producció de flors i fruits (Verkaik i Espelta 2006, Rodríguez-Calcerrada et al. 2008).

Estratègies de recuperació de les espècies vegetals

Les plantes tenen dues grans estratègies de resposta davant del foc, la germinació i la rebrotada, les quals donen lloc a quatre tipus d'espècies segons la seva estratègia de recuperació postincendi (Pausas et al. 2004a). Les espècies germinadores obligades són aquelles capaces de germinar i establir-se després del foc a partir de propàguls resistents al foc; mentre que les espècies rebrotadores obligades són aquelles que rebroten a partir d'estructures llenyoses que sobreviuen al foc. Les espècies facultatives combinen ambdues estratègies. I, finalment, algunes espècies no són capaces ni de germinar ni de rebrotar després del foc. La major o menor proporció d'aquests tipus d'espècies abans del foc determina en bona part la dinàmica de la regeneració de les comunitats forestals i, per tant, els canvis en la

composició de les comunitats (Keeley 1986, Arnan et al. 2007) i, a una escala més gran, del paisatge (Lloret et al. 2002, Moreira et al. 2011).

Patrons de regeneració i gestió forestal en escenaris postincendi

Després d'un foc, la resposta postincendi de comunitats mediterrànies dominades per espècies arbòries germinadores, com per exemple, per pi blanc (*Pinus halepensis*), acostuma a ser la germinació i establiment d'una elevada densitat de peus per hectàrea, implicant que aquestes masses de pi blanc presenten un estancament en el seu creixement individual per l'elevada competència pels recursos, així com un elevat risc d'incendi.

D'altra banda, s'ha descrit com, en determinades formacions forestals mediterrànies, a vegades, es dona una rebrotada molt vigorosa de determinades espècies arbòries o arbustives després del foc, les quals poden dominar clarament el procés de recuperació postincendi (Espelta et al. 2003). Això passa quan partim de comunitats forestals denses i clarament dominades per espècies rebrotadores ja abans del foc, com boscos dominats per espècies de *Quercus*, que rebroten de soca o capçada (Bonfil et al. 2004, Rodrigo et al. 2004, Arnan et al. 2007).

No obstant, això també pot succeir en boscos mixtes de rebrotadores i germinadores, on una espècie germinadora domina el vol del bosc i les rebrotadores poden formar un segon estrat arbori més baix. En aquest cas, la dominància de les espècies rebrotadores després del foc és conseqüència d'un establiment postincendi molt baix de l'espècie germinadora, ja sigui per condicions biòtiques (absència de llavors viables en el moment del foc) o abiòtiques adverses (condicions meteorològiques desfavorables), com pot succeir en boscos mixtes de *Pinus halepensis* o *P. pinaster* (Pausas et al. 2004a, Broncano et al. 2005) o per tractar-se d'espècies sense mecanismes de recuperació postincendi, com boscos de *P. pinea*, *P. sylvestris* o *P. nigra* (Ordóñez et al. 2004, Rodrigo et al. 2004, 2007; Pausas et al. 2008). En ambdós casos, les comunitats forestals passen a estar dominades per espècies de rebrotadores, sobretot d'espècies de *Quercus* que rebroten de soca després del foc (*Q. ilex*, *Q. faginea*, *Q. humilis* o *Q. x cerrioides*).

Aquestes bosquines de rebrotadores sovint presenten una estructura de capçada molt regular, amb una gran densitat d'individus per hectàrea i, a la vegada, amb un gran nombre de rebrots per individu. Aquest tipus d'estructura determina un estancament en el creixement individual, tant en longitud com en diàmetre, i una alteració de la seva reproducció, ja que hi ha una elevada competència pels

recursos, com l'aigua, la llum i els nutrients (Espelta et al. 2003). Aquesta estructura també representa un alt risc de propagació de nous incendis, ja que tenen una elevada càrrega de combustible vegetal, amb una gran continuïtat horitzontal i vertical (Oliveras et al. 2009, Fernandes 2009, Álvarez et al. 2012). Aquests boscos tenen menor valor paisatgístic i alhora, tampoc permeten cap aprofitament. Per una banda, són difícils de transitar i per tant, no es poden aprofitar com a pastures. I per altra, el creixement lent i la gran densitat de rebrots de diàmetres petits dificulta l'aprofitament forestal per llenya o fusta.

En aquest context, hi ha una gran necessitat per definir quin tipus de gestió forestal postincendi cal aplicar en les masses denses i regulars d'espècies germinadores o rebrotadores, amb la finalitat de garantir una estructura forestal més irregular, accelerar el seu creixement en diàmetre i alçada, millorar la seva regeneració natural i reduir el risc d'incendi. Alguns treballs ja han abordat aquest tema i han analitzat la resposta de la vegetació a l'aplicació de determinats tractaments silvícoles postincendi. Quan l'espècie dominant és germinadora es sol aplicar una aclarida d'individus, passant per exemple, de 50.000 a 1.000-1.500 peus/hectàrea per a *Pinus halepensis* (De las Heras et al. 2004, González-Ochoa et al. 2004, Verkaik i Espelta 2006, Moya et al. 2009, Piqué et al. 2011). Mentre que quan es tracta d'espècies rebrotadores, s'aplica una selecció de rebrots. En zones dominades per l'alzina (*Quercus ilex*) i el roure (*Quercus x cerrioides*), hi ha treballs que mostren com una selecció de rebrots de certa intensitat provoca un augment en el creixement dels rebrots reservats, tant en longitud com en diàmetre (Espelta et al. 2003, 2007) i una discontinuïtat del combustible (Cañellas et al. 2004). Ara bé, aquests tractaments s'apliquen en superfícies relativament grans i a tots els individus d'aquestes espècies presents a la zona, el que implica una reducció de la competència entre els rebrots d'un mateix individu, però també una reducció global a nivell de tota la parcel·la per l'ús dels recursos, especialment de la llum i l'aigua. Això dificulta la interpretació dels mecanismes que estan operant en aquestes respostes, perquè no es pot discernir entre els efectes de la reducció de la competència intraindividual i interindividual. D'altra banda, aquesta selecció de rebrots pot provocar una rebrotada induïda pel propi tractament de la tallada, la qual pot ser bastant vigorosa segons l'espècie i la intensitat del tractament, podent arribar a tenir un efecte negatiu sobre el creixement dels rebrots reservats (Ducrey i Turrel 1992, Riba 1998, Pelc et al. 2011). Tot i així, no totes les espècies rebrotadores mostren unes dinàmiques de creixement, de rebrotada induïda, d'autocompetència o de reproducció similars, pel que encara manca molta informació de com gestionar bosques dominades per altres espècies de rebrotadores.

Un altre tipus de tractament que s'acostuma a aplicar per reduir el risc d'incendi, sobretot en el perímetre de zones urbanes, és la desbrossada del sotabosc, ja sigui tant en boscos dominats per espècies germinadores com per espècies rebrotadores, ja que la finalitat és eliminar la continuïtat horitzontal de combustible, així com treure competència als arbres, afavorir la regeneració a través de l'establiment de noves plàntules o millorar l'accés a les zones forestals (Garolera 1991, CPF 1992, Mesón i Montoya 1993). En aquest cas, els efectes de l'aplicació d'aquest tractament han estat menys estudiats, probablement per ser un tipus de gestió més destinat a la prevenció d'incendis que no pas a la millora de l'estructura de les comunitats forestals.

En aquest context, els treballs de gestió forestal aplicats en escenaris postincendi es poden entendre com una segona pertorbació en l'ecosistema, els quals poden afectar el creixement i la reproducció de les espècies vegetals que s'han desenvolupat després del foc, i també poden afectar la fauna que s'ha establert a la zona cremada i dificultar la seva recuperació (Apigian et al. 2006). Per tant, els efectes d'aquests tractaments poden implicar un balanç d'efectes positius i negatius sobre els diferents processos que operen en els ecosistemes.

Efectes del foc i de la gestió forestal sobre les comunitats faunístiques

Els canvis que provoca el foc i/o la gestió forestal sobre l'estructura i la composició d'espècies vegetals d'una comunitat sovint poden repercutir en canvis en la seva composició faunística (Bawa i Seidler 1998, Vasconcelos 1999a, Bengtsson et al. 2000, Parr et al. 2004, Arnan et al. 2006, Lindenmayer et al. 2006, Nakamura et al. 2007, Parr i Andersen 2008, Andersen et al. 2009, Gómez i Abril 2011).

Els efectes del foc sobre la fauna poden ser directes, provocant la mortalitat o la fugida dels organismes, o bé indirectes, modificant el seu hàbitat. Així, espècies que sobreviuen al foc o retornen després de l'incendi poden tenir dificultats per viure en les noves condicions, ja que sovint s'ha alterat l'estructura de l'hàbitat, modificant-se les característiques ambientals, la disponibilitat de llocs de nidificació o les seves fonts d'aliment. Alhora, hi ha espècies que es poden establir en aquests nous hàbitats, amb unes característiques diferents a les d'abans de la pertorbació. Un exemple és l'aparició d'espècies d'aus d'hàbitats oberts, les quals són capaces de colonitzar zones cremades gairebé immediatament després del foc; i algunes de les quals – que habiten en matolls- fins i tot poden persistir quan, anys després,

aquestes àrees tornen a desenvolupar una estructura de bosc adult (Herrando et al. 2002, 2003; Pons i Bas 2005, Jacquet i Prodon 2009, Clavero et al. 2011).

D'altra banda, els aprofitaments forestals provoquen un canvi important de la coberta vegetal, implicant variacions en els recursos disponibles (radiació solar, humitat, temperatura, nutrients del sòl, etc.), de manera que la fauna associada es pot veure afectada per aquests canvis. S'ha comprovat que aquest efecte es pot traduir en una disminució de la diversitat faunística dels ecosistemes forestals (Thiollay 1997, Whitman et al. 1998, Davis et al. 2001, Lewis 2001, Sverdrup-Thygeson i Ims 2002). No obstant, aquests efectes depenen de la freqüència i la intensitat dels tractaments aplicats (Kavanagh i Webb 1998, Simon et al. 2000). Per tant, és recomanable valorar sempre que s'aplica una gestió forestal, que implica, per tant, una pertorbació del sistema, quin és l'efecte que pot tenir sobre la fauna de la zona.

S'han emprat diversos grups faunístics com a indicadors per avaluar els impactes de pertorbacions sobre els ecosistemes (Thiollay 1997, Kavanagh i Webb 1998, Davis et al. 2001, Andersen et al. 2009). Un d'aquests grups que ha estat emprat amb més èxit són les formigues, ja que són molt sensibles als canvis de l'ambient (Andersen 1997, 2000; Andersen i Majer 2004). Les formigues es troben involucrades en nombrosos processos ecològics clau (Hölldobler i Wilson 1990, Folgarait 1998, Gómez et al. 2003, Del Toro et al. 2012), com el cicle de nutrients del sòl (Lal 1988), el flux d'energia (Haines 1978), l'estructura i la fertilitat del sòl (Alvarado et al. 1981), la dispersió de llavors (Levey i Byrne 1993, Retana et al. 2004, Arnan et al. 2012), o bé relacions simbiòtiques amb animals i plantes (Hölldobler i Wilson 1990). A més a més, són fàcilment mostrejables, abundants en termes de biomassa i són el grup d'insectes socials més ric en espècies i ecològicament més divers, amb unes 12.500 espècies descrites en 290 gèneres que pertanyen a 21 subfamílies, tot i que aquesta xifra podria arribar a la de 25.000 espècies (Hölldobler i Wilson 1990, Lach et al. 2010). Tot això fa que s'hagin proposat i utilitzat amb èxit com a bioindicadores per analitzar impactes derivats de diverses pertorbacions (Hoffmann i Andersen 2003), com el foc (Parr et al. 2004, Arnan et al. 2006, Parr i Andersen 2008), inundacions (Adis 1982, Milford 1999, Ballinger et al. 2007), pastura (Bestelmeyer i Wiens 1996, Hoffmann 2000), mineria (Majer et al. 1984, Andersen et al. 2003), o processos d'urbanització (Lessard i Buddle 2005); i també per valorar l'efecte de la gestió forestal (Vasconcelos 1999a, Nakamura et al. 2007, Andersen et al. 2009, Gómez i Abril 2011).

L'arboç

L'arboç (*Arbutus unedo*, L., Ericaceae) és una espècie perennifòlia amb un baix grau d'esclerofil·lia (Gratani i Ghia 2002a), que pot assolir fins a 12 metres d'alçada (Sealy i Webb 1950). Rebrotta vigorosament després d'una pertorbació, com una tallada o un foc, i presenta un creixement lent. La seva àrea de distribució se situa a l'oest i sud d'Europa, nord d'Àfrica, Orient Mitjà i Macaronèsia, estant àmpliament distribuït per la Península Ibèrica i les Illes Balears. Normalment es troba com a espècie acompanyant en alzinars, sureres i rouredes; però també en pinedes, en estepars i bruguerars; habita en climes no excessivament freds ni secs.

L'arboç pot desenvolupar un elevat número de rebrots per soca i la mortalitat derivada de la competència entre els propis rebrots es produeix de forma més lenta que altres espècies de *Quercus* mediterrànies (Canadell et al. 1991). Aquests dos elements fan que l'arboç compti amb un número superior de rebrots per soca respecte espècies de *Quercus* durant el procés de regeneració de la comunitat, implicant una major limitació en el seu creixement i un elevat risc d'incendi (Figura 1).



Figura 1. L'elevada capacitat de rebrotada d'arboç fa que les seves bosquines assoleixin grans densitats de rebrots per individu i per hectàrea.

Pel que fa a la seva reproducció, floreix a la tardor i presenta panícules penjants, cadascuna amb 40-50 flors blanques (Sealy i Webb 1950), les quals són

pol·linitzades principalment per abelles relativament grans, com *Apis mellifera* i abellots del gènere *Bombus* (Herrera 2004). Produeix fruits esfèrics vermells entre 7 i 30mm de diàmetre, que poden contenir fins a 40 llavors en el seu interior (Sealy i Webb 1950, Moro 1995). Els fruits triguen un any a madurar, de manera que a la tardor-hivern es poden observar flors i fruits alhora en un mateix individu (Galán et al. 1998). Tot i així, el cicle reproductiu de l'arboç no ha estat mai clarament ben definit, degut a la complexitat de les seves diferents fases, trobant-se informació contradictòria en la bibliografia (Castro-Díez et al. 2003, Ogaya i Peñuelas 2007).

L'arboç presenta una variació en la producció de fruits entre anys (Herrera 1998), tot i que no prou extrema com per poder parlar pròpiament d'un fenomen d'anyivoria. També s'ha descrit una gran variació a nivell interindividual. Així, un individu pot produir entre 2,6 i 10kg de fruits/any (Gomes i Canhoto 2009, Molina et al. 2011), els quals actuen com a una important font d'aliment per a la fauna, especialment per ocells i mamífers (Arianoutsou i Diamantopoulos 1985, Rost 2011).

La gestió de les bosquines d'arboç en zones cremades

Evidències recents demostren que s'està produint un augment de la superfície coberta per bosquines d'arboç a Catalunya. Concretament, s'ha observat un augment de la seva superfície en el Mapa de la Cartografia d'Hàbitats de Catalunya respecte inventaris anteriors (Ministerio de Agricultura 1980, Departament de Medi Ambient i Habitatge i Universitat de Barcelona 2005), i aparentment sembla que estan sorgint en àrees afectades per una gran freqüència de focs (Figura 2). L'augment d'aquestes formacions, així com el fet que la informació de com gestionar l'arboç és molt escassa, ja sigui en zones de bosc adult quan es troba com a espècie acompanyant, o bé en un escenari de rebrotada després d'un foc, fa rellevant l'estudi de la resposta d'aquestes bosquines a l'aplicació de diferents tractaments forestals. Cal remarcar que no és evident que els mateixos tractaments recomanats per l'alzina i el roure rebrotats en zones cremades (selecció de rebrots amb més o menys intensitat, Espelta et al. 2003, Bonfil et al. 2004) donin una resposta equivalent en quant al seu creixement, rebrotada induïda o bé la reproducció de l'espècie, ja que l'arboç té una dinàmica de reducció de rebrots per soca molt més lenta degut a la pròpia competència intraindividual (Canadell et al. 1991). Així, per obtenir un número de rebrots baix, els rebrots que cal tallar en el cas de l'arboç hauria de ser molt més gran, el que podria provocar un major alliberament de recursos i una major reducció de l'ombra, i per tant, una rebrotada induïda pel

tractament molt més vigorosa que comprometria el creixement dels rebrots reservats (Castell i Terradas 1995).



Figura 2. Exemple de bosquines dominades per arboç (*Arbutus unedo* L.) desenvolupades a Súria (Bages) després dels incendis de 1983 i 1994.

Objectius

Els objectius generals d'aquesta tesi són, per una banda, definir quin és el paper dels incendis forestals en la formació de bosquines dominades per arboç a Catalunya. I, per l'altra, analitzar la resposta d'aquestes bosquines i la seva biodiversitat faunística, a l'aplicació de tractaments silvícoles, com són la selecció de rebrots i la desbrossada del sotabosc, realitzada en una zona cremada a Catalunya (NE Península Ibèrica), on l'arboç és l'espècie dominant en el vol del bosc. Per abordar aquests objectius la tesi s'ha estructurat en 4 capítols (Figura 3).

En el **primer capítol** s'analitza en quina mesura la recurrència dels incendis forestals i el tipus de coberta forestal anterior al foc, incrementa la presència de bosquines dominades per l'arboç a Catalunya. Per aconseguir-ho seguim dues aproximacions metodològiques: per una banda, s'utilitza un anàlisi SIG (Sistema d'Informació Geogràfica) per determinar el tipus de vegetació existent abans dels focs, així com el règim de recurrència de focs, en aquelles zones on s'han desenvolupat bosquines d'arboç. Per altra banda i per mitjà de treball de camp, analitzem com la manca de capacitat de recuperació després del foc de l'espècie arbòria dominant en pinedes de pi blanc dona lloc a l'aparició d'aquest tipus de bosquines.

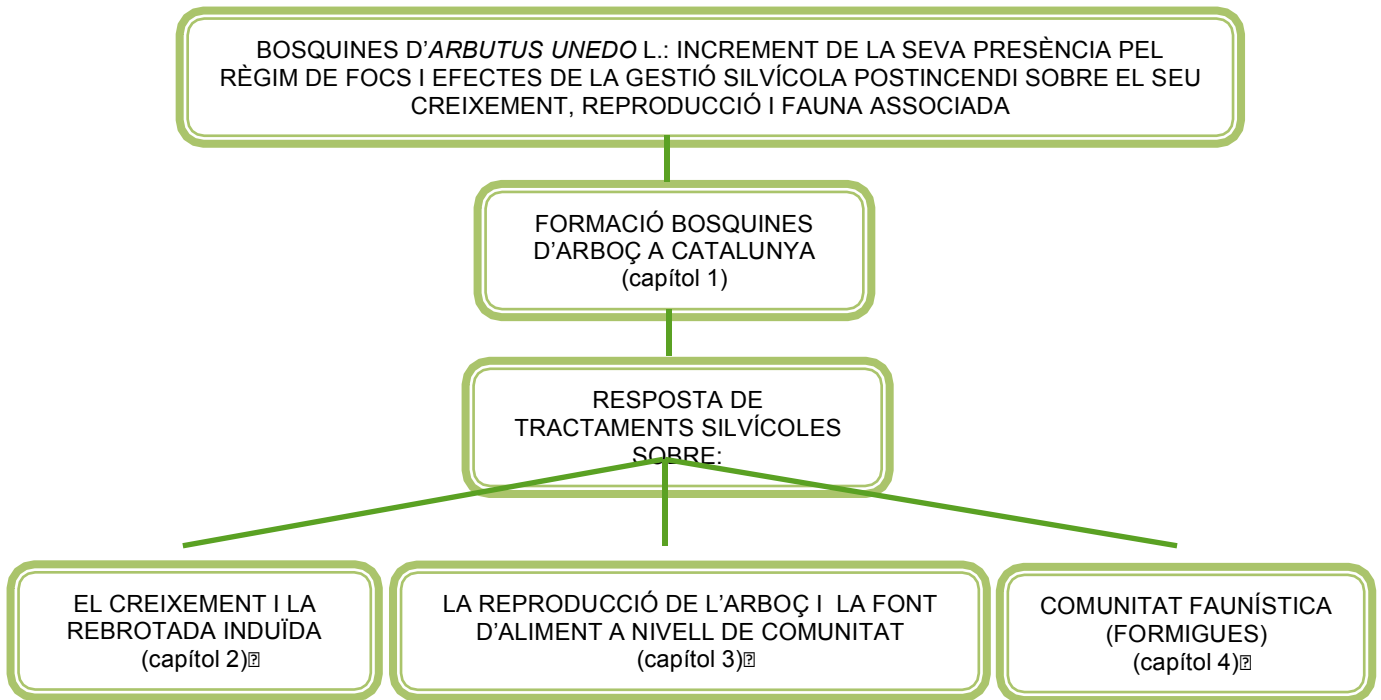


Figura 3. Esquema dels objectius de la tesi.

En el **segon capítol** s'estudia la resposta del creixement i rebrotada induïda de l'arboç després de l'aplicació de dues intensitats de tractament (selecció de rebrots i selecció de rebrots acompanyada de desbrossada del sotabosc) sobre els individus d'arboç de les bosquines regenerades després d'un incendi en una zona cremada a Catalunya (Figura 4). Aquesta resposta s'analitza tant a nivell d'individu com de parcel·la, separant els efectes de la reducció de la competència intraindividual i interespecífica, a través d'un disseny experimental de tractaments creuats.

En el **tercer capítol** explorem si aquests tractaments afecten també a la capacitat de producció de flors i fruits dels individus d'arboç, considerant la resposta tant a nivell de producció individual, de la qualitat dels fruits i la producció a nivell de bosquina. Això ens permet valorar les implicacions d'aquests tractaments en la capacitat reproductora de l'arboç, però també les repercussions a nivell de la comunitat faunística associada, ja que els fruits d'arboç són una font d'aliment molt important per a la fauna en aquestes bosquines.

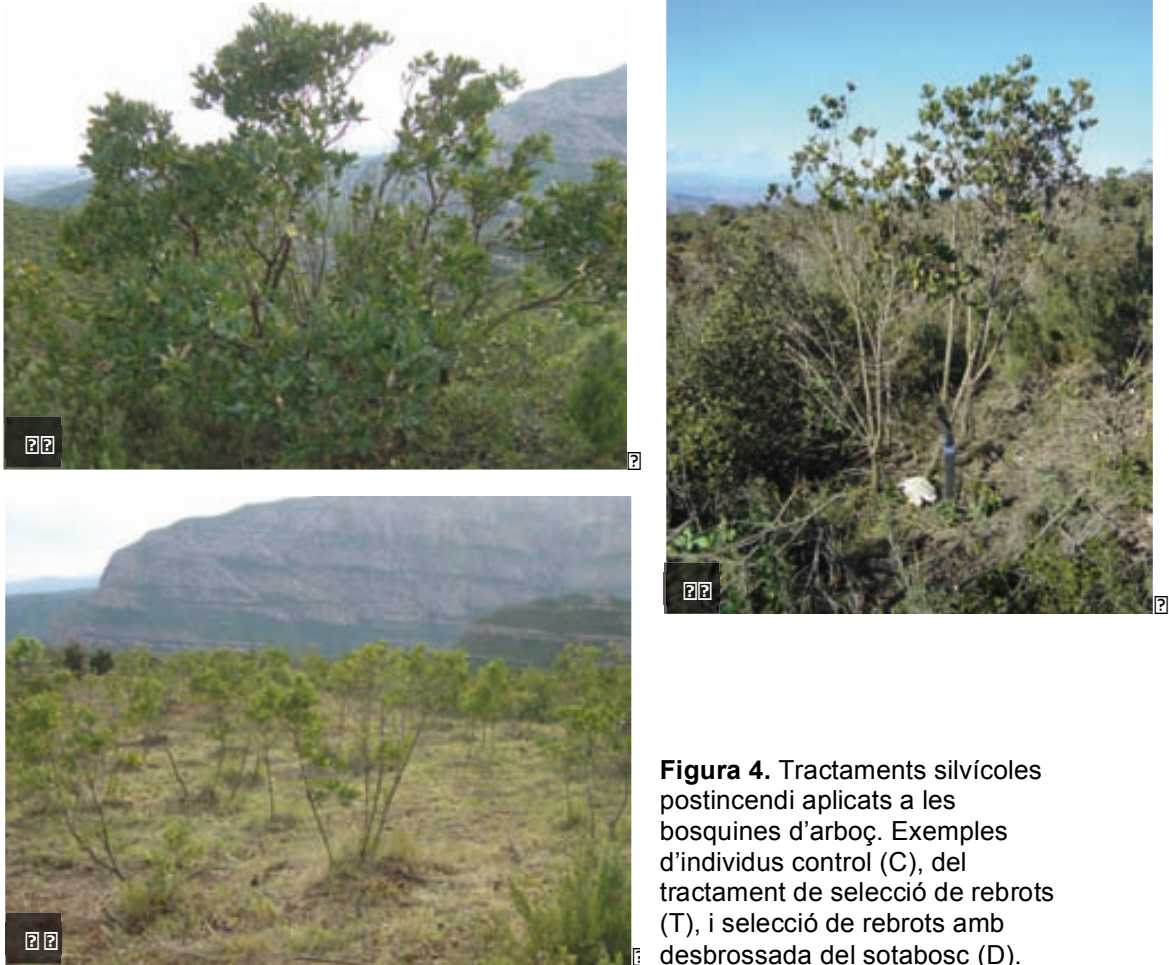


Figura 4. Tractaments silvícoles postincendi aplicats a les bosquines d'arboç. Exemples d'individus control (C), del tractament de selecció de rebrots (T), i selecció de rebrots amb desbrossada del sotabosc (D).

En el **capítol quart** analitzem si aquests tractaments de selecció de rebrots afecten a la comunitat faunística d'aquestes bosquines, en concret, a les comunitats de formigues. Així, ens plantegem si: a) els tractaments silvícoles postincendi en bosquines d'arboç afecten l'estructura i composició de les comunitats de formigues, ja que es modifiquen les condicions ambientals; i b) si una intensitat major del tractament implicarà major afectació de les comunitats de formigues.

Àrea d'estudi

Aquesta tesi s'ha dut a terme en dues àrees d'estudi, una que inclou l'àrea de distribució de l'arboç a Catalunya, i que correspon al primer capítol; i una altra, Esparreguera, on es van establir unes parcel·les experimentals que es van seguir durant 5 anys i es va analitzar la resposta de l'aplicació d'uns tractaments silvícoles postincendi (selecció de rebrots, acompanyada o no de la desbrossada del

sotabosc) sobre el creixement i la reproducció de l'arboç, així com el seu efecte sobre la comunitat de formigues, emprades com a bioindicadores, dels que se'n deriven els estudis corresponents al segon, tercer i quart capítol d'aquesta tesi, respectivament.

L'àrea d'estudi del primer capítol comprèn tota l'àrea de distribució d'*Arbutus unedo* a Catalunya (Figura 5). Aquesta àrea s'ha emprat pels anàlisis amb GIS i ha estat delimitada utilitzant el Banc de Dades de Biodiversitat (BioCat 2010). Comprèn 2.000.000 ha i inclou regions bioclimàtiques oceàniques pluviestacionals mediterrànies i oceàniques temperades submediterrànies (Worldwide Bioclimatic Classification System 1996-2009).

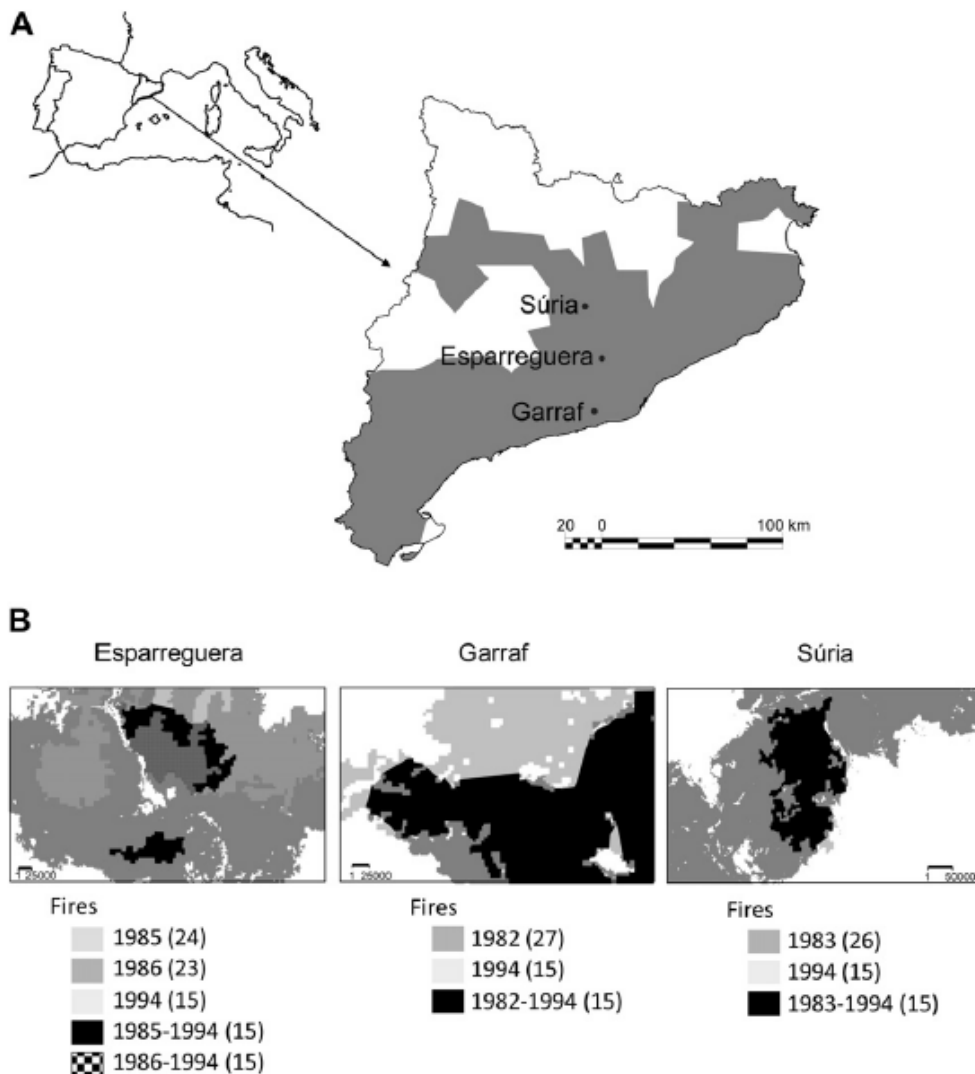


Figura 5. A) Mapa de Catalunya indicant l'àrea d'estudi del primer capítol de la tesi (en gris), l'àrea de distribució d'*Arbutus unedo* L., i les tres localitzacions d'estudi (punts negres); B) Mapes detallats dels llocs d'estudi per mitjà de treball de camp, indicant amb colors els diferents incendis ocorreguts; en parèntesi, temps transcorregut entre l'últim incendi forestal i el treball de camp.

Les bosquines d'arboç van ser detectades durant els darrers inventaris forestals de Catalunya (Departament de Medi Ambient i Habitatge i Universitat de Barcelona 2005), mentre que eren gairebé inexistents en inventaris forestals previs (Ministerio de Agricultura 1980). El treball de camp d'aquest capítol s'ha dut a terme en tres localitats dins l'àrea de distribució de l'arboç: Garraf (41° 18' N, 1° 49' E), Esparreguera (41° 35' N, 1° 52' E) i Súria (41° 50' N, 1° 47' E).

D'altra banda, l'àrea d'estudi de la resta de capítols se situa a Esparreguera, prop del Parc Natural de la Muntanya de Montserrat (41° 35' N; 1° 52' E, Catalunya, nord-est de la Península Ibèrica), en una àrea afectada per diversos incendis forestals en les últimes dècades (1985, 1986 i 1994, amb 223, 1.335 i 3.165 hectàrees cremades, respectivament) (Figures 5 i 6). Aquesta zona es troba entre els 390 i 500m sobre el nivell del mar. Presenta un clima mediterrani sec subhúmit (segons l'Índex d'humitat de Thornthwaite), amb una temperatura mitjana anual de 13,5°C i una precipitació mitjana anual de 650-700mm (Ninyerola et al. 2003). Abans d'aquests incendis, aquesta zona tenia una coberta de bosc dominat per pi blanc (Ministerio de Agricultura 1980). Posteriorment als tres incendis, es va desenvolupar un mosaic en la composició de les cobertes forestals. A les zones cremades una vegada s'hi va desenvolupar una coberta dominada per pi blanc. Les zones cremades dues o tres vegades van ser dominades per rebrotadores: ja sigui alzina o bé masses mixtes d'alzina i roure (78 hectàrees), o masses d'arboç (238 hectàrees), essent en aquesta darrera zona on s'han desenvolupat els capítols 2, 3 i 4 d'aquesta tesi (Figura 6). En aquesta zona es van establir unes parcel·les de mostreig de les que es va fer un seguiment durant 5 anys. Si bé el disseny experimental aplicat es detalla a cadascun dels capítols, cal indicar que en el nostre disseny no s'ha inclòs la desbrossada del sotabosc com a tractament propi perquè aquest estudi es cenyeix als efectes de tractaments forestals reals aplicats pels gestors i, normalment, la desbrossada del sotabosc s'acostuma a emprar com a tractament complementari als treballs d'aclarida d'individus o bé de selecció de rebrots.

D'altra banda, l'aplicació de la selecció de rebrots no es va fer seguint la pràctica més habitual emprada pels gestors i l'administració, sobretot en el cas de l'alzina i el roure, que consisteix en tallar els rebrots amb motoserra, deixant únicament entre un i tres rebrots en peu (Vericat et al. 2011). Per una banda, es va valorar que l'arboç, en ser una espècie que compta amb una rebrotada molt vigorosa després d'un foc o una tallada (fins a 60 rebrots/soca, 12 anys després del foc; Lúdia Quevedo, observació personal), la reducció fins a tan pocs rebrots, com es fa per a l'alzina i el

roure, podria provocar deixar rebrots molt desprotegits i causar una elevada mortalitat dels rebrots reservats. D'altra banda, la motoserra no és una eina adient per a la tallada de rebrots de diàmetre petit, ja que provoca la tremolor del rebrot en el moment de la tallada i en haver encara tants rebrots, fàcilment pot causar ferides en els rebrots reservats, que poden provocar la mortalitat del rebrot reservat (Lídia Quevedo, observació personal). Per això, per aplicar els tractaments als arboços vam establir com a criteri tallar un de cada cinc rebrots, deixant els més sans, rectes i de major diàmetre, per tal que els individus quedessin compensats estructuralment i evitant deixar els individus grans amb molt pocs rebrots. A més, la tallada es va fer amb tisora elèctrica, evitant, per tant, produir ferides en els rebrots reservats. Aquest criteri ha permès garantir la total supervivència dels rebrots reservats.



Figura 6. Zona d'estudi dels capítols dos, tres i quatre de la tesi, situada a Esparreguera, prop del Parc Natural de la Muntanya de Montserrat.

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CHAPTER 1

Forest fire occurrence increases the distribution of a scarce forest type in the Mediterranean Basin

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Forest fire occurrence increases the distribution of a scarce forest type in the Mediterranean basin

Abstract/Resum

Here we report how fire recurrence increases the distribution of a scarce forest type in NE Spain that is dominated by the resprouter tree species *Arbutus unedo*. We used a combination of GIS and field surveys to determine the effect of fire and pre-fire vegetation on the appearance of *A. unedo* forests. In the field, we also analyzed the factors that promote fire and lead to the appearance of *A. unedo* forests. Our results reveal an increased occurrence of *A. unedo* forests in NE Spain in recent years; this phenomenon was strongly related to fire recurrence and the vegetation type present prior to fire. Most *Pinus halepensis* forests that burned more than once gave rise to *A. unedo* forests. Our results indicate that these conversions were related to a reduction in pine density coupled with increases in the density and size of *A. unedo* trees due to recurrent fires. Given that fires are increasing in number and magnitude in the Mediterranean, we predict a major change in landscape structure and composition at the regional scale.

Keywords: *Arbutus unedo*, forest dominance, GIS analysis, Mediterranean, *Pinus halepensis*, wildfires

En aquest treball constatem com la recurrència d'incendis incrementa la distribució d'un tipus de bosc escàs al NE d'Espanya, que està dominat per l'espècie arbòria rebrotadora *Arbutus unedo*. Vàrem utilitzar una combinació de SIG i mostrejos de camp per determinar l'efecte del foc i de la vegetació existent en el moment del foc, en l'aparició dels boscos d'*A. unedo*. En el treball de camp, també vam analitzar els factors que promouen el foc i donen lloc a l'aparició dels boscos d'*A. unedo*. Els nostres resultats revelen un increment de l'ocurrència dels boscos d'*A. unedo* al NE d'Espanya en els darrers anys; aquest fenomen estava fortament relacionat amb recurrència dels focs i el tipus de vegetació present abans del foc. Bona part dels boscos de *Pinus halepensis* que es van cremar més d'una vegada van donar pas als boscos d'*A. unedo*. Els nostres resultats indiquen que aquestes conversions estaven relacionades amb una reducció de la densitat de pins combinat amb increments en la densitat i mida dels arboços, degut als focs recurrents. Donat que els incendis forestals estan augmentant en número i magnitud al Mediterrani, preveiem un canvi més important en l'estructura i composició del paisatge a una escala regional.

1. Introduction

Fire is undoubtedly one of the main factors determining the current forest landscape of the Mediterranean Basin (Naveh, 1975; Pausas et al., 2008; Trabaud, 1987). The effect of fire on Mediterranean ecosystems is variable and depends on the intensity, area, and recurrence of the disturbance (Broncano and Retana, 2004; Eugenio and Lloret, 2004; Paula and Ojeda, 2006; Pausas et al., 2003; Tsitsoni, 1997). In recent decades, the fire regime in the Mediterranean Basin has changed, and is now characterized by larger, more intense wildfires (Lloret et al., 2009; Moreira et al., 2011; Pausas, 2004; Piñol et al., 1998). The effect of these changes in fire regime on the forest landscape of the Mediterranean basin will depend on the resilience of different forest types (Rodrigo et al., 2004).

Mediterranean-type plant communities have high resilience after fire (Hanes, 1971; Trabaud and Lepart, 1980) because many species regenerate by sprouting after top-kill (López-Soria and Castell, 1992), while others, despite adult mortality, regenerate through the germination of fire-protected seeds stored in the soil or in the canopy bank (Keeley, 1986; Lloret, 1998). The sprouting strategy maintains populations through the persistence of individuals that were present before the fire and, therefore, in densities similar to or lower than before the fire (Pausas et al., 2004), depending on sprouting success (Quevedo et al., 2007). In contrast, the persistence of seeder populations is based on the substitution of burned and killed plants by new individuals that can reach much higher densities in the initial stages of post-fire regeneration than before the fire (Pausas et al., 2004). The sprouting ability of most Mediterranean woody species is generally very high (López-Soria and Castell, 1992; Paula et al., 2009; Quevedo et al., 2007) and is not affected by an increase in fire recurrence (Bonfil et al., 2004; Cruz et al., 2003; Riba, 1998), except when recurrence is extremely high (Delitti et al., 2005). In contrast, a higher fire recurrence might reduce the ability of seeder species to regenerate after fire when the interval between fires is less than the time required for individuals to reach the reproductive stage, a phenomenon known as immature risk (Zedler, 1995). This is the case for *P. halepensis* when fire occurs at intervals of less than 10-20 years, which is the time required for this species to effectively rebuild its canopy seed bank after fire (Verkaik and Espelta, 2006). In addition to these fire-adapted species, there are also Mediterranean species incapable of regenerating after fire (Arnan et al., 2007; Paula et al., 2009; Retana et al., 2002; Rodrigo et al., 2004).

A slow recovery of plant cover in forests where the dominant tree species do not regenerate after crown fire has been reported in the Mediterranean and around the world (Arianoutsou et al., 2010; Arnan et al., 2007; Chen et al., 2009; Cohn et al., 2011; Eugenio and Lloret, 2004). This could allow not only greater opportunities for colonization of open spaces but also a shift to forest covers dominated by resprouter species that were already present in the understory (Barton, 2002; Fulé & Covington, 1998; Ganatsas et al., 2012; González-Tagle et al., 2008; Retana et al., 2002; Rodrigo et al., 2004). On the one hand, newly created open areas would mainly be colonized by species with fire-resistant seed banks or opportunistic species whose seeds are dispersed from neighboring unburned habitat (Hanes, 1971; Ne'eman et al., 1992; Ojeda et al., 1996). The effectiveness of the latter mechanism depends largely on seed dispersal distance as well as the size of the burned area. Moreover, most sprouting Mediterranean woody species are efficient long-distance dispersers that rely on vertebrates for seed transport (Gomez, 2003; Herrera, 1995, 1998; Jordano and Schupp, 2000; Pons and Pausas, 2008), even in burned areas (Rost et al., 2009). In spite of this efficiency, it seems that these species seldom successfully colonize recently burned areas (Quevedo et al., 2007) because they need some vegetative cover to maintain the minimum levels of humidity essential for seedling establishment (Mesleard and Lepart, 1991; Zavala et al., 2000). On the other hand, mixed evergreen and deciduous oak coppices in NE Spain have become more abundant in areas affected by large wildfires over the last two decades. This process has been promoted by the failure of the area's dominant pine species to regenerate, irrespective of fire occurrence, coupled with the resprouting of *Quercus* species previously present in the forest understory (Broncano et al., 2005; Kazanis et al., 2011; Proença et al., 2010; Rodrigo et al., 2004; Silva et al., 2011). It is even more striking that the species now composing the new forest cover were seldom previously described as dominant; instead, they were mainly considered to be understory companion species.

The aim of this study is to analyze the extent to which fire recurrence increases the distribution of a scarce forest type, specifically forests dominated by the resprouter tree species *Arbutus unedo* L. in Catalonia (NE Spain). To this end, we employed a two-part approach. First, we used GIS to separate out the effects of fire versus pre-fire vegetation type on the appearance of these forests. Second, in the field, we analyzed how the vegetative response to fire mediates the appearance of these forest types when post-fire recovery of the dominant species has failed. Given that most *A. unedo* forests were found in *Pinus halepensis* forests that had burned more

than once (see Results), we compared *P. halepensis* forests that had burned once (i.e. areas where post-fire pine recovery may or may not have failed) with *P. halepensis* forests that had burned twice within a relatively short time period (i.e. areas where post-fire pine recovery had likely failed). We analyzed whether the appearance of this scarce forest type was due to: a) the understory species becoming structurally dominant (i.e. pine disappearing and *A. unedo* resprouting and occupying more space, thus becoming the dominant species) and/or b) the failure of *P. halepensis* to recover following fire, with new *A. unedo* individuals from off-site potentially colonizing newly created open areas.

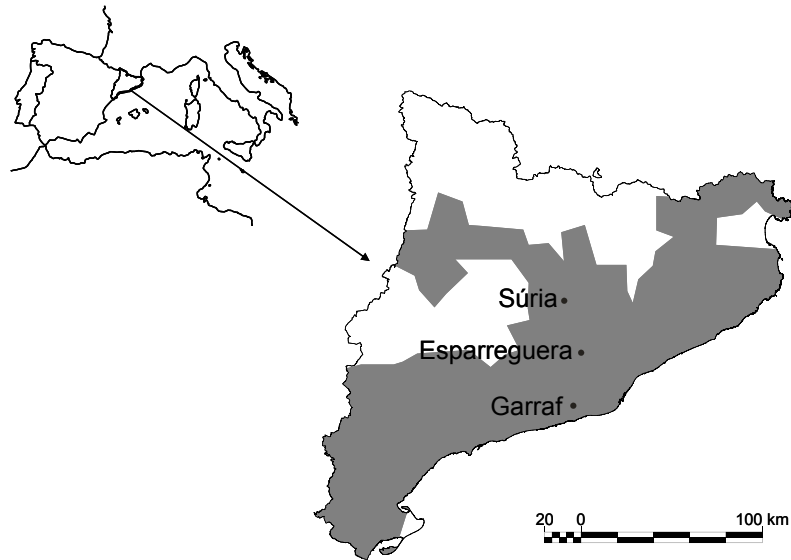
2. Material and methods

2.1. Study area

This study was conducted within the distribution range of *A. unedo* (including both where it occurred in the understory and tree cover) in Catalonia (Figure 1). This area was identified using the Biodiversity Data Bank (BioCat, 2010). It comprises 2,000,000 ha and encompasses Mediterranean pluviseasonal oceanic and temperate oceanic sub-Mediterranean bioclimatic regions (Worldwide Bioclimatic Classification System, 1996-2009).

Arbutus unedo forests were found during the latest forest inventories of Catalonia (Departament de Medi Ambient i Habitatge and Universitat de Barcelona, 2005), whereas they were almost nonexistent in previous forest inventories (Ministerio de Agricultura, 1980). This species is relatively common in the understory of most typical Mediterranean forests (Galán et al., 1998; Gratani and Ghia, 2002). At first glance, the distribution of these new forests coincides with some of the areas burned in the last few decades.

A



B

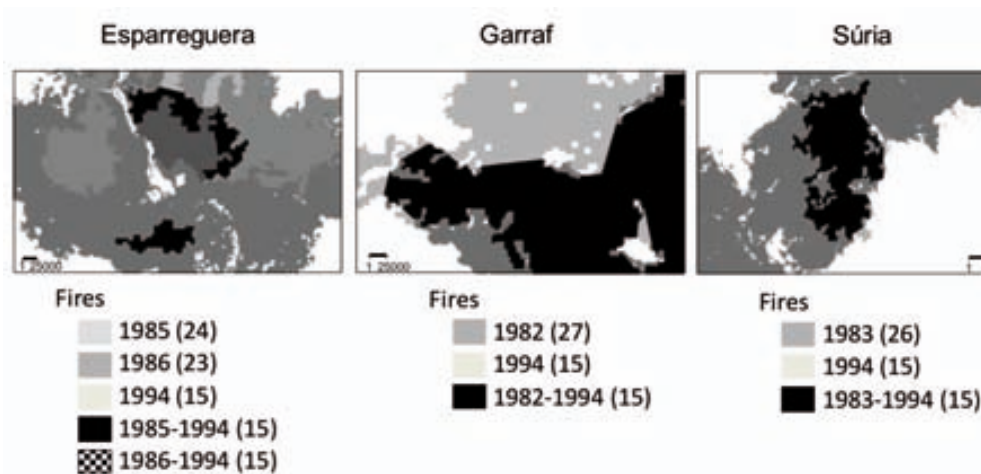


Figure 1. a) Map of Catalonia indicating the overall study area (in grey), i.e. the distribution area of *A. unedo*, and the particular study sites (black circles); b) Detailed maps of the study sites depicting the different wildfires; in brackets, time elapsed between the last fire and field sampling.

2.2. The effect of fire and land cover prior to disturbance on the appearance of *A. unedo* forests

Using the GIS MiraMon package (Pons, 2008), we analyzed the presence of *Arbutus unedo* forests in the study area in 1996 relative to a) wildfires that occurred between 1980 and 1996; and b) land cover type before these fires (i.e., 1979). For this analysis, we extracted and overlaid different maps (Figure 2). First, we obtained a map of the study area that included the distribution of *A. unedo* forests in 1996 (hereafter, MA96) by selecting all habitats categorized as *A. unedo* forests from the Mapa de la Cartografia d'Habitats de Catalunya (Departament de Medi Ambient i Habitatge and Universitat de Barcelona, 2005); the field survey for this map was conducted in 1996. Second, we generated a map that displayed the fires that occurred in Catalonia between 1980 and 1996 in the same area (hereafter, MF96) using the Forest Fire Map of Catalonia 1975-2010 (Institut Cartogràfic de Catalunya, 2011). Finally, we extracted a map that showed the distribution of different land cover types within the study area prior to the fires (i.e. 1979) (hereafter, MC79) from the Mapa de Cultivos y Aprovechamientos (Ministerio de Agricultura, 1980). These land covers were grouped into five categories: scrublands, oak forests, *P. halepensis* forests, forests of other conifers, and other land cover. Urbanized surface was not considered in our analyses.

First, we overlaid the map of the distribution of the *A. unedo* forests (MA96) on the map of fire occurrence (MF96) (Figure 2). This process enabled us to generate a new map (MAF96) that showed the distribution of *A. unedo* forests in unburned areas, once-burned areas, and multiply burned areas, and then compute the percentage of area occupied by *A. unedo* forests in each of these categories. Second, we overlaid this new map (MAF96) on the land cover map (MC79) (Figure 2). Since we also had information on fires that occurred from 1975 to 1980, the areas that had burned once between 1975 and 1980 and again between 1980 and 1996 were regarded as twice-burned. We thus generated a final map (MAFC96) that categorized different land cover types according to their fire frequency and showed whether or not they became *A. unedo* forest (Figure 2). From there, we calculated the percentage of *A. unedo* forest area associated with each pre-fire land cover type and fire category.

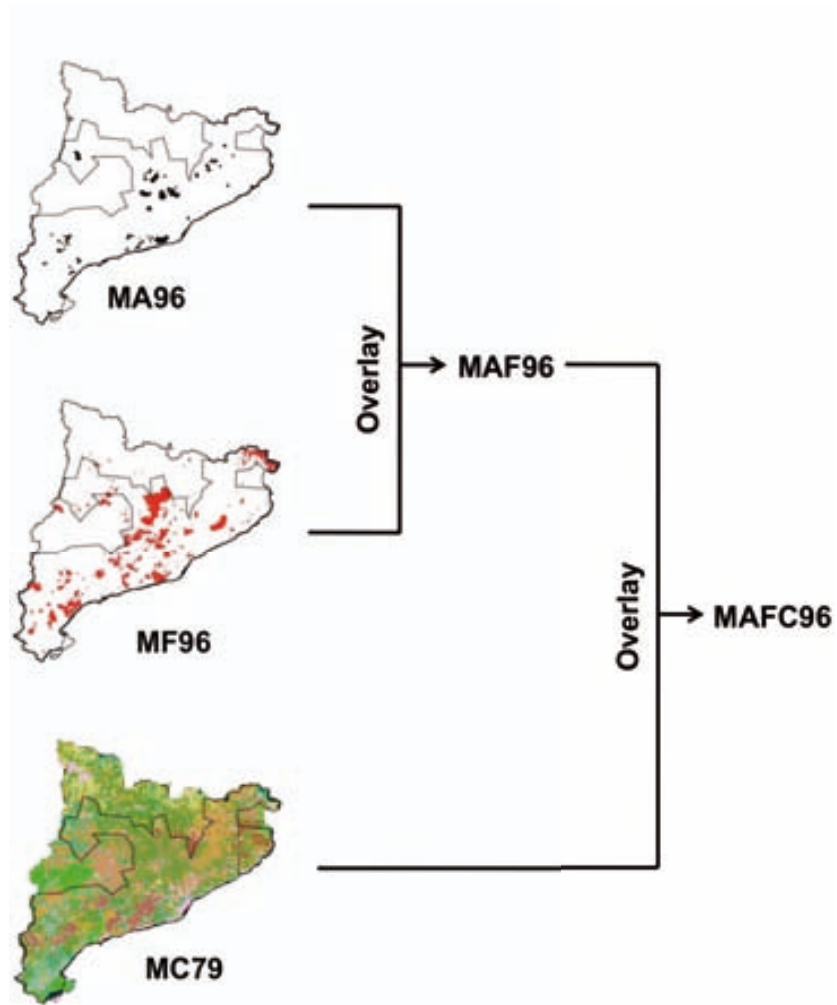


Figure 2. Schematic representation of the map overlay procedure used to analyze the relationship between *A. unedo* forest occurrence, wildfires, and pre-fire land cover types in Catalonia. The dotted line indicates the distribution range of *A. unedo* as a companion species and thus defines the area in which the analyses were carried out. Abbreviations: MA96, distribution area of *A. unedo* forests in 1996; MF96, distribution of fires that occurred from 1980 to 1996; MC79, distribution of the different land cover types prior to fire (i.e., 1979); MAF96, overlay between MA96 and MF96; and MAFC96, overlay between MAF96 and MC79.

2.3. Appearance of *A. unedo* forests in burnt *P. halepensis* stands

The second stage of the study was carried out at three different sites within the study area described above; the sites represented a gradient from coast to inland: Garraf (1° 49' North 41° 18' East), Esparreguera (1° 52' North 41° 35' East), and Súria (1° 47' North 41° 50' East) (Figure 1). All share a typical Mediterranean dry sub-humid climate. These locations had all repeatedly burned in the last three decades, and, prior to the first fire, had been characterized by mature *Pinus halepensis* forests that included *Arbutus unedo* in the understory. Garraf has a mean annual rainfall of 550-600 mm and a mean annual temperature of 14-15°C. We selected two study sites there. The first had burned once as a consequence of a 5,638-ha wildfire in 1982. The second had burned during the same 1982 fire and again in 1994 (4,594 ha affected). A total of 3,158 ha burned during both events. Esparreguera has a mean annual rainfall of 647 mm and a mean annual temperature of 15.4°C. The entire study site had burned in a 1994 wildfire that covered 3,104 ha; other fires had taken place previously in 1985 (223 ha) and 1986 (1,383 ha). As a consequence, a mosaic of once-burned and multiply burned areas was created. We only selected study sites that had burned once in the 1994 wildfire or that had also previously burned in either the 1985 or 1986 wildfires. Súria has a mean annual rainfall of 650-700 mm and a mean annual temperature of 12-13°C. The entire study area was disturbed by a large wildfire that burned 38,262 ha in 1994; however, a 1983 fire had already burned 611 ha, and 580 ha were burned by both fires. Because some burned areas at all locations were subject to post-fire management, they were excluded from our sampling regime.

In 2009, we established a total of 123 10-m radius circular plots that were spread over the three study sites: 53 in Esparreguera (28 in once-burned areas and 25 in twice-burned areas), 27 in Garraf (13 in once-burned areas and 14 in twice-burned areas), and 30 in Súria (15 in both once- and twice-burned areas). Plots were selected so as to be similar in elevation, aspect, slope and pre-fire plant composition, in spite of any differences in fire recurrence (once- or twice-burned), thus limiting variability. However, we could not control for all the differing factors that could potentially be linked to fire recurrence. In each plot, we counted the number of *P. halepensis* and *A. unedo* individuals taller than 0.5 m, as well as those belonging to other resprouter species (*Quercus ilex* and *Q. cerrroides*) that share the forest understory with *A. unedo* and might affect its post-fire regeneration via competition. We also counted the number of sprouts for each *A. unedo* individual.

From the total densities the relative density of each species (the percentage of the total density it accounts for) was calculated and the dominant species, i.e. the species with the highest relative density, was identified for each plot. A species had to have a relative density that exceeded 50% in order to be regarded as a dominant; consequently, plots where no species exceeded this value were removed from the analysis. The number of once-burned and twice-burned plots that each species dominated were compared for each study site with a Chi-square test; species identity comprised the rows and fire recurrence status (once- or twice-burned) the columns.

In order to explore if fire mediates the appearance of *A. unedo* forests, the effect of fire recurrence on *A. unedo* was analyzed by fitting general mixed linear models with the *lme* function of the R statistical package (R Development Core Team, 2010). The density of *A. unedo* individuals and the number of sprouts per individual were included in two separate models as dependent variables. Fire recurrence (once- or twice-burned) was included as a fixed factor, and original site identity was included as a random effect. The density of *A. unedo* individuals and the number of sprouts per individual were respectively square-root and log-transformed to approximate normality. Since there was a strong random effect found in the analysis of *A. unedo* individual density (see results), i.e. site accounted for much of the variability, we reanalyzed the data in order to see if different patterns were present at different sites. We thus analyzed the effects of fire recurrence (once- or twice-burned), site (Garraf, Esparreguera, or Súrria), and their interaction on the individual density of *A. unedo* by means of two-way ANOVAs. The replication units were the plots. To understand the patterns found for *A. unedo* density, we also analyzed the effects of fire recurrence on the stem density of *P. halepensis* and the individual density of other important resprouters (*Q. ilex* and *Q. cerrroides*) using two-way ANOVA tests. Variables were square-root or log-transformed when necessary to approximate normality.

3. Results

3.1. The effect of fire and land cover prior to disturbance on the appearance of *A. unedo* forests

Our GIS analyses (Figure 2) show that the *A. unedo* forests observed in 1996 occupied land previously covered by *P. halepensis* forests and, to a lesser extent, scrublands, oak forests, and other cover types (Table 1). More interestingly, these *A. unedo* forests were mainly found in once-burned areas as compared to unburned areas and areas that had burned more than once (Table 1). In areas that burned more than once, *A. unedo* mainly occurred in tracts of land formerly characterized by *P. halepensis* forests (Table 1). The pattern was the same for once-burned areas, although *A. unedo* also occurred on former scrublands and areas with other cover types (Table 1).

Table 1. Absolute and relative values of the area covered by different vegetation types that were replaced by *A. unedo* forests by 1996. Note that the size of the areas we examined were quite different depending on their fire history (1,676,452 ha for unburned areas, 153,060 ha for areas burned once, and 3,239 ha for areas burned more than once from 1980 to 1996) and cover category (344,996 ha for *P. halepensis* forests, 207,802 ha for other conifer forests, 991,169 ha for oak forests, 203,506 ha for scrublands, and 977,278 ha for other cover types).

	Unburned	Burned once	Burned more than once	Overall
<i>P. halepensis</i> forests	220.5 (3.1%)	2,842.6 (39.6%)	924.7 (12.9%)	3,987.8 (55.6%)
Forests of other conifers	34.4 (0.5%)	91.1 (1.3%)	0.2 (0.0%)	125.7 (1.8%)
Oak forests	739.1 (10.3%)	0.0 (0.0%)	0.0 (0.0%)	739.1 (10.3%)
Scrublands	331.5 (4.6%)	1,125.4 (15.7%)	124.8 (1.8%)	1,581.7 (22.1%)
Other cover types	224.6 (3.1%)	346.3 (4.8%)	166.4 (2.3%)	737.3 (10.2%)
Overall	1,550.1 (21.6%)	4,405.4 (61.4%)	1,216.1 (17.0%)	7,171.6 (100.00%)

It is notable that 4.5% of the area that burned more than once from 1980 to 1996 was covered by *A. unedo* forests in 1996; this compares markedly with the 2.5% of *A.*

unedo cover in the once-burned area. Both these results contrasted greatly with the mere 0.1% of the unburned area that became *A. unedo* forest by 1996.

When examining pre-fire land cover in greater detail, we found that over 70% of *P. halepensis* forests that had been burned more than once became *A. unedo* forests (Figure 3). In contrast, when this forest type only burned once or not at all, it was rarely replaced by *A. unedo*, less than 5% of the time. About 20% of other cover types and 14% of scrublands that burned twice or more became *A. unedo* forests. When they remained unburned or burned only once, they rarely became *A. unedo* forests. The appearance of *A. unedo* forests in areas previously characterized by oak or other conifer forests was insignificant, regardless of fire history (Figure 3). However, these areas also rarely burned from 1980 to 1995 (at most, 0.1% of the total of each land cover type).

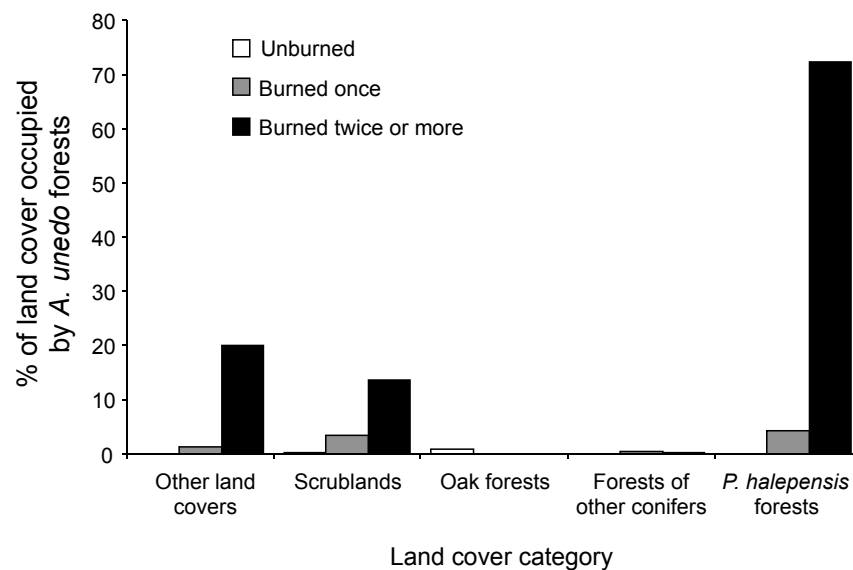


Figure 3. Percentage of each land cover category (scrublands, oak forests, *P. halepensis* forests, forests of other conifers, and other cover types) in 1979 that had become *A. unedo* forest by 1996, classified according to fire history between 1980 and 1995 (unburned, burned once, and burned twice or more).

Our analysis also revealed that around 20% of *A. unedo* forests in 1996 occurred in unburned areas (Table 1). Specifically, these *A. unedo* forests were found in locations that used to be oak forests and scrublands and, to a lesser extent, *P. halepensis* forests, other cover types, and other conifer forests (Table 1). Although

we cannot rule out that *A. unedo* invaded these areas or that a small number of individuals already present at these sites could have multiplied rapidly and successfully, neither possibility is likely given the low recruitment rate of this species under normal conditions (Pérez-Ramos and Marañón, 2012). Instead, these results suggest that there was an error in the classification survey of the Mapa de Cultivos y Aprovechamientos (Ministerio de Agricultura, 1980), and that *A. unedo* forests were probably mistakenly classified as scrubland or oak forest. This would mean that *A. unedo* forests already existed in 1979. However, the fact that they were improperly classified in 1979 and that a low percentage of unburned areas were classified as *A. unedo* forests in 1996 (0.1%, see above) suggest that they were nonetheless very scarce in 1979. Indeed, it would seem that *P. halepensis* forests (the vegetation type most often replaced by *A. unedo* forest, see above) were properly classified.

3.2. Appearance of *A. unedo* forests in burned *P. halepensis* stands

When we examined tree density in the field, the most abundant tree species in all three sites was *Pinus halepensis*, with 3310 ± 487 (mean \pm SE) trees/ha, followed by *Arbutus unedo* (686 ± 75 trees/ha), and two sparse resprouter species: *Quercus ilex* (170 ± 28 trees/ha) and *Q. cerrioides* (116 ± 22 trees/ha). The relative abundance of minor species varied among the sites (Figure 4).

Fire recurrence had a significant effect on the proportion of plots dominated by different species (e.g., the species that had the highest relative tree density) in Esparreguera ($\chi^2=18.3$, $df=2$, $p=0.0001$) and Súrria ($\chi^2=22.5$, $df=1$, $p<0.0001$), but not in Garraf ($\chi^2=0.96$, $df=1$, $p=0.326$). *P. halepensis* dominated all once-burned areas. While *A. unedo* dominated twice-burned areas at two of the study sites (Esparreguera and Súrria), *P. halepensis* remained dominant in Garraf in such plots (Figure 5). This difference in dominance was especially noticeable in Súrria.

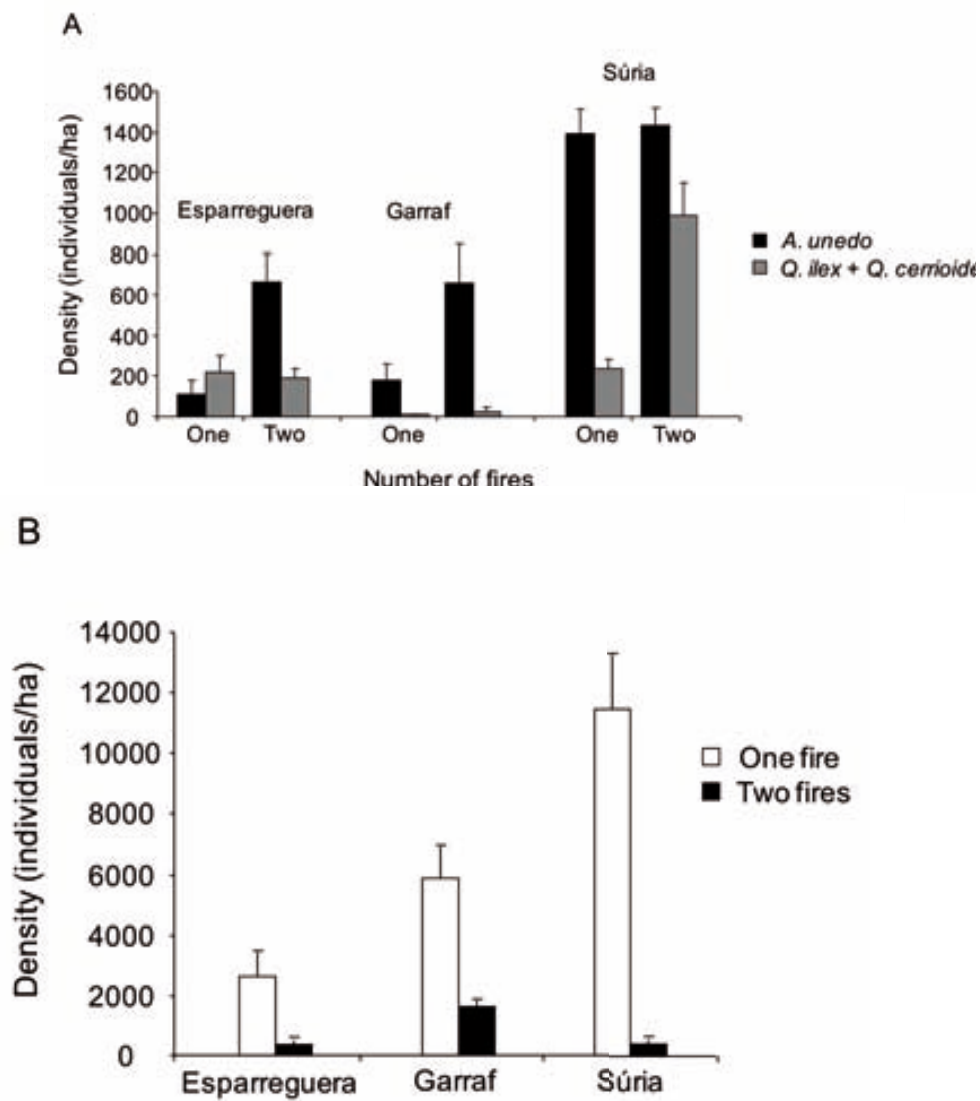


Figure 4. Mean \pm S.E. individual density (number of individuals/ha) per study site and fire recurrence category of a) *Arbutus unedo* and other resprouter species (*Quercus ilex* and *Q. cerroides*) and b) *Pinus halepensis*.

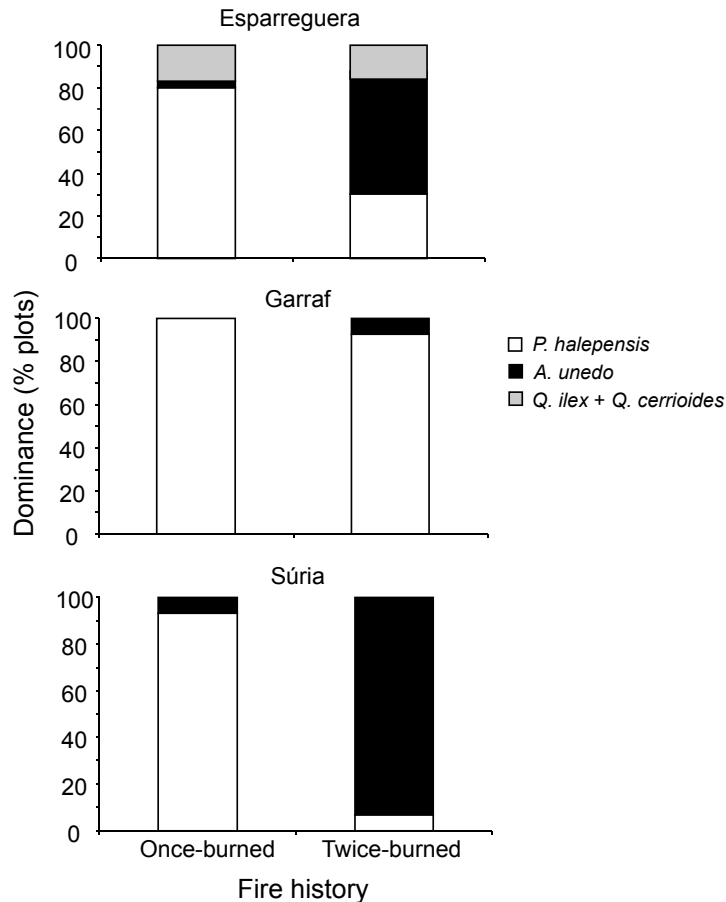


Figure 5. Dominance (% of plots where a given species had the highest relative tree density) of *P. halepensis*, *A. unedo*, and oak species in once- and twice-burned areas at each study site.

Fire recurrence had a strong effect on the density of *A. unedo* individuals (mixed models estimate=10.8±2.3, $t=4.7$, $df=106$, $P<0.0001$) and on its number of sprouts per individual (mixed models estimate=0.25±0.04, $t=6.1$, $df=81$, $P<0.0001$). Both measures were higher in plots that burned twice (913±114 individuals/ha and 11.4±0.6 sprouts/individual) than in plots that burned once (468±91 individuals/ha and 8.0±0.8 sprouts/individual). However, there was also a strong random effect of site in the analysis of density, as shown by the comparison of models with and without the random factor (L.Ratio= 53.1, $p<0.0001$). This comparison was not significant in the analysis of the number of sprouts per individual ($p=0.216$). We then reanalyzed the data in order to explore deviance from the global pattern of *A. unedo* density among sites. Two-way ANOVAs indicated that there was a significant effect of the interaction between the number of fires and site on *A. unedo* density ($F_{2,104}=3.9$, $p=0.02$), such that density increased following two fires at two of the sites (Esparreguera and Garraf), but no increase was detected for Súria (Figure 4a).

Fire recurrence also impacted the other species potentially competing for space with *A. unedo*. The individual density of *P. halepensis* ($F_{2,104}=4.8$, $p=0.010$) was affected by an interaction between fire recurrence and site: it was always reduced when two fires had occurred, with the reduction being more pronounced in Súrria (Figure 4b). For oaks, this interaction was not significant ($F_{2,104}=1.7$, $p=0.187$), but the main effects of fire recurrence ($F_{1,104}=4.6$, $p=0.03$) and site ($F_{2,104}=23.3$, $p=0.0001$) were significant. Oak individual density was higher in twice- versus once-burned areas at all three sites; it was also higher in Súrria than in Garraf and Esparreguera (Figure 4a).

4. Discussion

In this work, we describe the appearance of *A. unedo* forests where they were previously not present in NE Spain. Our results strongly suggest that their advent is the result of an increased incidence of wildfires. Almost 80% of these forests occurred in burned areas. Furthermore, 7% of the burned area was characterized by *A. unedo* forests, whereas only 0.1% of the unburned area was covered by this forest type. Other studies have demonstrated that fires change Mediterranean Basin forest cover (Baeza et al., 2007; Ganatsas et al., 2012; Piussi, 1992; Retana et al., 2002; Rodrigo et al., 2004; Silva et al., 2011) and, consequently, the landscape. These changes mainly took place in forests where the predominant pine species faced regeneration failure, independent of fire, and whose understories were dominated by resprouter species (particularly *Quercus* species). These species occupy a greater proportion of the vegetative cover when they resprout after fire. Similar forest replacement events have been reported elsewhere in the world, where anomalously large, intense, or frequent crown fires have favored resprouters over obligate seeders, a consequence of seeders failing to establish post-fire (Barton, 2002; Chen et al., 2009; Cohn et al., 2011; Fulé and Covington, 1998; González-Tagle et al., 2008; Rebertus et al., 1994; Russell-Smith et al., 1998). These forest cover changes increase the range of forests that are comprised of resprouter species, which were already able to form forests themselves. More importantly, fire can also generate novel forest cover types or, more commonly, enlarge the distribution range of scarce forests, as we have shown in the case of *A. unedo*. This species has previously been considered a companion species in Mediterranean forests (Galán et al., 1998; Gratani and Ghia, 2002; Konstantinidis et al., 2006) and has rarely been described as forming forests. It is worth noting that, although this forest type was virtually

absent in our study area, it is more common in the western Iberian Peninsula (Perez, 1976).

Following wildfire, *A. unedo* forests most often replaced *P. halepensis* forests, scrublands, and other land cover types (Table 1, Figure 3). These land cover changes were mainly mediated by fire recurrence (Figure 3), although such changes were also seen in once-burned areas. The recovery of vegetation in areas where fire recurs may be even more delayed than in areas that burn only once (Eugenio and Lloret, 2004), thus generating more open spaces that can be exploited by species that were non-dominant prior to fire or that arrive from off-site. *Pinus halepensis* is an obligate seeder that depends on its canopy seed bank to recover following fire (Arianoutsou and Ne'eman, 2000; Habrouk et al., 1999; Herranz et al., 1997; Retana et al., 2002; Tapias et al., 2001). It requires sufficient time between fires to adequately replace itself after a new fire event. If time between fire events is too short, then the post-fire regeneration of this species will be unsuccessful (immature risk; Moya et al., 2008; Zedler, 1995). This suggests that seeder species dominate the multiply burned areas in which *A. unedo* forests arise, namely scrublands and those of other cover types. In contrast, oak species are good at resprouting following fire; more than 90% of burned individuals survive (López-Soria and Castell, 1992; Rodrigo et al., 2004). Indeed, their resprouting abilities are not impacted by disturbance frequency (Bonfil et al., 2004). Thus, forests dominated by resprouter species are not expected to be affected by fire recurrence, and species replacement, such as by *A. unedo*, appears unlikely. However, we lacked the sampling strength to test this prediction; although our results show virtually no replacement of oak forests by *A. unedo* forests, fire occurrence in these forest types was almost nil. Overall, pre-fire forest cover type and fire recurrence seemed to determine the appearance of *A. unedo* forests, suggesting that these changes depend on how well dominant species regenerate and, consequently, the creation of open spaces (Arnan et al., 2007), areas with little vegetative cover where other tree species might establish themselves. Our results indicate that *P. halepensis* forests were those most frequently converted to *A. unedo* forests, a process promoted by fire recurrence.

If we focus on *P. halepensis* forests (Table 1, Figure 3), we could ask how the response of vegetation to fire mediates the replacement of recurrently burned *P. halepensis* stands by *A. unedo* forests. Our results suggest that the answer may lie in a reduction of *P. halepensis* density coupled with an increase in the density and tree size of *A. unedo* that results from recurrent fires (Figure 4). The time interval required

for canopy seed bank recovery may be between 10 and 20 years for *P. halepensis* (Daskalakou and Thanos, 1996; Eugenio et al., 2006; Verkaik and Espelta, 2006), whereas the time interval between fires was less than 15 years at all three of our study sites (11, 12, and 8 or 9 in Súrria, Garraf and Esparreguera, respectively). Consequently, the low density of *P. halepensis* in twice-burned areas (Figure 4b) was likely due to an insufficiently long fire interval; trees regenerated after the first fire did not have enough time to refresh the canopy seed bank. In contrast, regeneration in once-burned areas stemmed from a well-developed canopy seed bank that reproductive pines had established. Furthermore, the seed dispersal distances of *P. halepensis* are relatively short, less than ~50m (Nathan et al., 2000). Since the wildfires considered here were of considerable size and almost entirely canopy fires, regeneration through seed rain from unburned areas was essentially irrelevant on our study sites. It should be noted that *P. halepensis* can also fail to regenerate after a single fire, as evinced by the fact that a small percentage of these pine forests that burned once were also replaced by *A. unedo* forests (Figure 3); this small percentage comprised most of the area that gave rise to *A. unedo* forests (Table 1). Although it is generally accepted that *P. halepensis* successfully regenerates after a single fire, there may be some variability at the local (Pausas et al., 2004) and regional scale. This could be due to adverse climatic conditions, which can reduce the canopy seed bank before fire (Espelta et al., 2011) or kill newly germinated seedlings via drought after fire.

The relatively higher density of *A. unedo* individuals in areas that burned twice (Figure 4a) would require this species to not only maintain but also greatly increase effective population size in the face of recurrent fires. Indeed, *A. unedo* has been described as a typical resprouter species (López-Soria and Castell, 1992), allowing it to maintain similar levels of density across the burned areas, regardless of fire number (Pausas et al., 2004); however, if frequent fire diminishes their resprouting ability, their density might be slightly lower in areas that burn often. In contrast, we found that the density of *A. unedo* individuals was much higher in areas of higher fire recurrence. Although we did not test the mechanisms that could account for this pattern, we propose two hypotheses. On the one hand, there might be post-fire establishment of new *A. unedo* individuals. *Arbutus unedo* seeds can be dispersed over long distances through bird droppings (Mesleard and Lepart, 1991), and fleshy-fruit eating birds may be abundant in burned areas (Pons and Prodon, 1996; Rost et al., 2009). Furthermore, if *P. halepensis* fails to regenerate and herbaceous plants and shrubs do not recover in areas of highest fire recurrence (Eugenio and Lloret,

2004), open spaces could be created and maintained for long periods of time. In these spaces, competition may be less fierce for *A. unedo* seedlings than in once-burned areas, where *P. halepensis* seedling densities may be high in open microhabitats just after fire (Arianoutsou and Ne'eman, 2000; Broncano et al., 2005; Daskalidou and Thanos, 2004). Thus, *A. unedo* individuals would arrive as seeds from off-site (Rost et al., 2009) and their colonization would be concentrated in spaces where *P. halepensis* regeneration had failed. However, this hypothesis contradicts previous work that has described *A. unedo* as a tree species that establishes in areas of dense vegetation and thus appears to be a mid-successional species (Mesleard and Lepart, 1991; Santana et al., 2011). It has also been reported that *A. unedo* density decreases with disturbance frequency (Santana et al., 2011).

An alternative hypothesis is that a seedling bank (sensu Pons and Pausas, 2006) is maintained in the forest understory. These seedlings would correspond to a pool of suppressed, slow-growing individuals that are waiting for improved conditions and thus might resprout after a fire. They would not develop after a single fire because the faster growth rate of *P. halepensis* seedlings would confer competitive superiority (Broncano et al., 2005). However, they could resprout after a subsequent fire because pine regeneration would fail; *A. unedo* seedlings could then develop and become successful adults. Further research is needed to determine the reason for the increased post-fire density of *A. unedo* individuals. Is *A. unedo* able to settle open areas in the early stages after fire, or is a seedling bank being maintained, or are both mechanisms operating? Whatever the mechanism, our work clarifies that certain resprouter tree species consistently increase their individual density under recurrent fire regimes, which suggests that the disturbance caused by fire is a significant force in the recruitment of new individuals for these species. Since we controlled for different factors that could introduce variability and thus obscure differences between fire recurrence factor levels, we can assume that *A. unedo* individual density was similar across plots before they burned. As a result, the recruitment ability of this species (and others similar) is intermediate to that of seeder species, which renew the entire population following fire, and that of the most vigorously resprouting species, which add few or no new individuals (Pausas et al., 2004). As new individuals of this species are rarely incorporated into undisturbed Mediterranean forests (Pérez-Ramos and Marañón, 2012), fire might serve as an important recruitment mechanism in these populations. Across all our study sites, *A. unedo* individuals had a higher number of resprouts in areas that burned twice versus those that burned once. This finding fits with others that show that the replacement of pine

forests by oak forests is linked to the failure of pines to regenerate post-fire and the vigorous sprouting of oaks (Barton, 2002; Retana et al., 2002; Rodrigo et al., 2004). We also found that the individual density of *A. unedo* was consistently higher in twice-burned versus once-burned areas. This pattern contrasts with past work, which did not find an increase in density following fire.

Both patterns, the decrease in the density of *P. halepensis* individuals and the increase in the density of *A. unedo* individuals, signify that fire brings about changes in species dominance in forests (Figure 5), resulting in the conversion of *P. halepensis* forests to *A. unedo* forests. An exception to this outcome was seen in Garraf, where both the decrease in *P. halepensis* density and the increase in *A. unedo* density (Figure 4) were too weak to trigger changes in species dominance (Figure 5). This is likely because the reduction in *P. halepensis* density on this site was not as marked as on the other sites, probably because the canopy seed bank recovered somewhat before the second fire. Meanwhile, in Esparreguera, the contrast between the two patterns was apparent (Figure 4), with one forest type clearly replacing the other. In contrast, although there was also a clear change of forest types, from pine to *A. unedo*, in Súrria (Figure 5), the individual density of *A. unedo* did not increase (Figure 4). The recruitment of new *A. unedo* individuals may have been limited by the high overall density of resprouters at this site (Figure 4), despite a significant decrease in the individual density of *P. halepensis*; perhaps no open space was left to be occupied. Two features of this study system - the incorporation of new individual oaks and the competitive impact of oaks on *A. unedo* - still remain unknown. Regardless, the increased individual density of oaks in twice-burned areas was not enough to exceed the values of *A. unedo*, which could proceed to dominate forest cover through vigorous sprouting.

4.1. Conclusions

Our results provide an additional example of forest types that do not follow the assumed auto-successional post-fire regeneration pattern (e.g., Arnan et al., 2007; Barton, 2002; Retana et al., 2002; Rodrigo et al., 2004; Russell-Smith et al., 1998; Silva et al., 2011), including those that usually demonstrate good post-fire recovery at maturity. This finding has important implications given that the number and magnitude of fires are increasing in the Mediterranean region (Pausas, 2004; Piñol et al., 1998), and it is predicted that fire risk will increase as a consequence of current climate warming (Moreira et al., 2011; Moriondo et al., 2006). More significantly, our

results also provide evidence that changes in fire regimes can give rise to new types of forest cover or increase the distribution range of scarce forests in a regional context. In other words, so-called companion species, which have been assumed to be restricted to the understory, may actually create their own forest formations depending on fire dynamics. These patterns might herald changes in the structure of the present landscape in the Mediterranean Basin.

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CHAPTER 2

Selective thinning of *Arbutus unedo* coppices following fire: Effects on growth at the individual and plot level

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Selective thinning of *Arbutus unedo* coppices following fire: effects on growth at the individual and plot level

Abstract/Resum

In recent years, several Mediterranean forests subject to fire are becoming increasingly dominated by the resprouter species *Arbutus unedo* L (strawberry tree). However, there is little information available about the management of these areas, and it is not clear if the approaches utilized for other, more abundant Mediterranean resprouter species such as *Quercus ilex* and *Q. cerrioides* would produce similar results for *A. unedo*. In this study, performed in the NE Iberian Peninsula, we analysed the effect of two post-fire treatment types, selective thinning and selective thinning plus understory clearing, on the growth of retained *A. unedo* resprouts and the new resprouting induced by the treatment itself. Treatment effects were analyzed at both the individual and plot level. Our results showed that, in the short term, retained resprouts on treated trees grew more in height and diameter (absolute and relative) than those on control trees, with no differences seen between treatment types. In the intermediate term, all the strawberry trees occurring on treated plots grew longer, and this growth was greater in plots that had been both thinned and cleared. New, induced resprouts were unaffected by the type of treatment applied. In contrast to other studies, we failed to find a negative relationship between the degree of induced resprouting and retained resprout growth. We therefore conclude that, irrespective of understory clearing, the selective thinning of *A. unedo* improves coppice vertical structure. Retained resprouts grow more in diameter and will thus more rapidly become exploitable as firewood and timber. Also, the tree is kept free of dead fuel, reducing the risk of spreading fire.

Keywords: selective thinning; induced resprouting; fire prevention; strawberry tree; Mediterranean.

En els darrers anys, diversos boscos mediterranis afectats pel foc estan esdevenint, cada vegada més, dominats per l'espècie rebrotadora *Arbutus unedo* L. (arboç). De totes maneres, hi ha poca informació disponible sobre la gestió d'aquestes àrees, i no està clar que els tractaments utilitzats per altres espècies rebrotadores mediterrànies més abundants, com ara *Quercus ilex* i *Quercus cerruoides*, produeixin resultats similars per *A. unedo*. En aquest estudi, dut a terme al NE de la Península Ibèrica, vam analitzar l'efecte de dos tipus de tractaments postincendi, selecció de rebrots i selecció de rebrots amb desbrossada del sotabosc, sobre el creixement dels rebrots reservats d'*A. unedo* i la nova rebrotada induïda pel propi tractament. Els efectes dels tractaments van ser analitzats a nivell individual i de parcel·la. Els nostres resultats van mostrar que, a curt termini, els rebrots reservats dels arbres tractats van créixer més en longitud i diàmetre (valors absolut i relatiu) que els dels arbres control, sense haver diferències entre tipus de tractament. A mitjà termini, tots els arboços de les parcel·les tractades van créixer més, i aquest creixement va ser superior a les parcel·les on s'havia aplicat la selecció de rebrots amb la desbrossada del sotabosc. Els nous rebrots induïts no van ser afectats pel tipus de tractament aplicat. A diferència d'altres estudis, no vam trobar una relació negativa entre el grau de rebrotada induïda i el creixement dels rebrots reservats. Per tant, concloem que, independentment de la desbrossada del sotabosc, la selecció de rebrots millora l'estructura de les bosquines d'*A. unedo*. Els rebrots reservats creixen més en diàmetre i per això poden esdevenir explotables més ràpidament com a llenyes i fusta. A més, l'arbre es manté lliure de combustible mort, reduint el risc de propagació del foc.

1. Introduction

Fire plays an important role in shaping the structure and composition of Mediterranean forest ecosystems (Moreira et al., 2011; Naveh, 1994; Piussi, 1992; Rodrigo et al., 2004; Trabaud, 1994). This role depends on fire regime characteristics, such as frequency, severity, and size, as well as on pre-fire forest composition (Pausas et al. 2008). Plants have two main strategies in response to fire, seeding and resprouting, which give rise to four functional groups. After a fire, obligate seeder species germinate and re-establish themselves from fire-resistant propagules, while obligate resprouter species resprout from resilient woody structures. Facultative species utilize both strategies, while some species are incapable of either seeding or resprouting post-fire (Paula et al. 2009; Pausas et al., 2004a). The abundance of these different functional groups prior to a fire will largely determine forest regeneration dynamics (Arnan et al. 2007; Rodrigo et al., 2004).

In some Mediterranean forests, particular shrub or tree species demonstrate extremely vigorous resprouting following fire and thus dominate the post-fire recovery process (Clemente et al. 1996; Espelta et al., 2003; Lloret and Vilà, 2003; Quevedo et al., 2007; Silva et al. 2011; Ganatsas et al. 2012). This phenomenon is seen in forests dominated by resprouters, such as *Quercus* species that resprout from the stump or crown (Arnan et al., 2007; Bonfil et al., 2004; Rodrigo et al., 2004). It can also occur, however, in mixed forests, where obligate tree seeder species or species without any post-fire recovery mechanisms dominate the forest canopy while resprouters form the understory. Resprouter species may become dominant following fire because seeder species show limited re-establishment, due to adverse biotic or abiotic conditions, as has been observed in forests of *Pinus halepensis* or *P. pinaster* mixed with *Quercus* species (Broncano et al., 2005; Eugenio and Lloret, 2004; Rodrigo et al., 2004). They may also become dominant if the fire has eliminated species that lack mechanisms for post-fire recovery (Ganatsas et al. 2008, 2012; Ordóñez et al., 2004; Pausas et al., 2008; Rodrigo et al. 2007, 2004; Vilà-Cabrera et al., 2012). These coppices of resprouters often have very regular canopy structure, characterized by high tree density and a large number of resprouts per tree. This

structure limits plant growth by generating intense competition for resources such as water, light, and nutrients (Espelta et al., 2003). Moreover, the accumulation of fuel and greater horizontal and vertical continuity at these sites greatly enhance the propagation risk of new fires (Álvarez et al. 2011; Fernandes, 2009; Oliveras et al., 2009). Such woodlands are also unsuitable for human use, as they are difficult to pass through and therefore cannot be used for grazing. Furthermore, their slow growth and high density of small-diameter resprouts rule out any commercial exploitation of firewood and timber.

In this context, it is essential to define the post-fire forest management strategy that is most appropriate for these coppices of resprouting species, so as to promote a more regular forest structure, enhance plant growth, and reduce fire risk. In areas dominated by *Quercus ilex* and *Quercus cerroides* previous studies have reported that selective thinning promotes the growth of retained resprouts, both in height and diameter (Espelta et al., 2003), and limits fuel accumulation (Cañellas et al., 2004). However, these treatments were applied to all the trees found in relatively large areas. Consequently, it is likely that thinning reduced both intraindividual competition (among an individual's resprouts) as well as interindividual competition for resources like light and water across the plot. Furthermore, selective thinning induces a new wave of basal resprouting, which, depending on the species and treatment intensity, can be quite vigorous and ultimately have a negative effect on retained resprout growth (Ducrey and Turrel, 1992; Pelc et al., 2011; Riba, 1998).

The strawberry tree, *Arbutus unedo* L., is a small tree that may grow up to several m tall (Moro, 1995), obligate resprouter species (Paula et al., 2009) that is largely restricted to the Mediterranean and Macaronesian area, but is found at some Atlantic locations in France and Ireland (Villar 1993). Although common in the understory of many Mediterranean forests (Galán et al., 1998; Gratani and Ghia, 2002), it is rarely a dominant species, except in some southern Iberian Peninsula mountains where some communities are dominated by *A. unedo* (Torres et al., 2002). In general, *A. unedo* is not the dominant species after fire, and its abundance in the post fire community depends of the competition with seeders and another resprouter species (Moreira et al., 2012; Clarke et al. 2012).

However, coppices dominated by *A. unedo* have appeared and are becoming increasingly abundant in burned areas in the NE Iberian Peninsula (Arnan et al., 2013). These coppices develop as follows: a pine forest, in which *A. unedo* is present in the understory, burns, the pines do not recover, and the strawberry tree resprouts vigorously, turning into the dominant tree species. *Arbutus unedo* resprouting generates a large number of resprouts per stump, and self-selection (or competition-mediated mortality) among those resprouts takes place more slowly (Canadell et al., 1991) than in Mediterranean *Quercus* species. These two factors account for *A. unedo*'s much higher number of resprouts per stump over the course of regeneration. They might also limit plant growth and augment fire risk. The horizontal and vertical continuity of fuel allows fires to endure for considerable periods of time. Data on the management of strawberry tree coppices are very scarce. It is not clear if *A. unedo* would react similarly to *Q. ilex* and *Q. cerroides* if managed similarly (Bonfil et al. 2004; Espelta et al. 2003). The intensity of selective thinning should be much higher and retain only a few resprouts, thus increasing resource availability and reducing the shade cast by the resprouts themselves. Such an approach could lead to much more vigorous treatment-induced resprouting, which would hinder the growth of retained resprouts (Castell and Terradas 1995). Furthermore, it is necessary to know if selective thinning at the individual level has any effect in the absence of local resource reduction.

In this study, we analysed the growth of *A. unedo* individuals in response to two thinning treatments. Selective thinning of resprouts and selective thinning of resprouts plus mechanical clearing of the surrounding vegetation were applied to *A. unedo* individuals in a burned area dominated by *A. unedo*. We used an experimental design crossing general plot treatment with treatments applied at individual level. This design allowed us to test for the effects of intraindividual versus interindividual (intraspecific) and interspecific competition. We evaluated the following hypotheses:

- (i) For individual trees, retained resprouts will increase in both height and diameter in response to the reduction in the number of resprouts per stump (reduction of intraindividual competition).

- (ii) At the plot level, the growth (diameter and height) of each individual's retained resprouts will be greater if resource availability on the plot as a whole is higher, whether because of selective thinning of other trees or mechanical clearing of vegetation (reduction of interindividual and interspecific competition).
- (iii) The degree of induced resprouting will be proportionally greater in the selective thinning plus clearing treatment than in the selective thinning treatment.
- (iv) The greater the degree of treatment-induced resprouting, the greater the negative effect on retained resprout growth will be.

2. Material and Methods

2.1. Study area

This study was undertaken in the north of the Baix Llobregat region (41° 35' N; 1° 52' E, Catalonia, NE Iberian Peninsula) (Figure 1). Elevation ranges from 390 to 500 m above sea level. The region has a subarid Mediterranean climate (according to Thornthwaite's scale), with a mean annual temperature of 13-14 °C and a mean annual precipitation of 650-700 mm (Ninyerola et al., 2003). The dominant soils are Calcic Haploxerepts associated to Typic Xerorthens (Soil Survey Staff, 2010) developed on sandstones and clays.

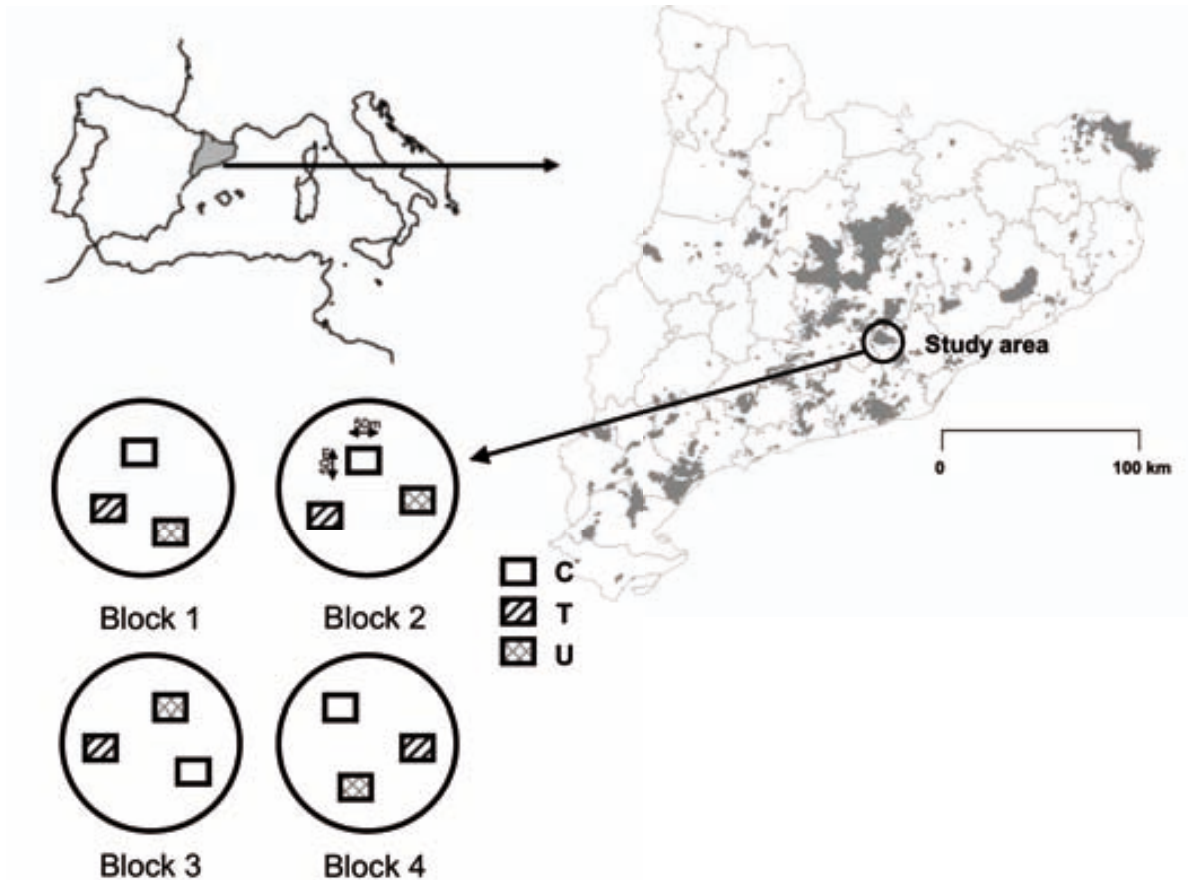


Figure 1. Geographical location of the study area in the north of the Baix Llobregat region, Catalonia (NE Iberian Peninsula). Grey patches indicate areas that burned from 1975 to 2002. The distribution of the experimental treatments within each block is also shown (control plots, C; plots with selective thinning, T; and plots with selective thinning plus understory clearing, U).

The area has burned several times in recent years (1985, 1986, and 1994, with 223, 1335, and 3165 hectares burned, respectively). Before these fires occurred, this area was covered by forests dominated by Aleppo pine (*Pinus halepensis* Mill.) (Mapa de Cultivos y Aprovechamientos, 1980). Afterwards, a mosaic of land covers developed. In the areas that burned once, tree cover is still dominated by (regenerated) Aleppo pine. The areas that burned two or three times have become dominated by resprouters, mainly *Q. ilex* and *Q. cerrioides* (covering 78 ha), or the strawberry tree (*Arbutus unedo* L.) (covering 238 hectares), which is the focus of this study. Strawberry trees are most often

located in sunny areas, where they vigorously resprout after fire (up to 60 resprouts per stump) (Lidia Quevedo, unpublished data).

2.2. Experimental design and application of post-fire experimental treatments

In 2006, we established four sampling areas (hereafter, blocks) in a burned area SW aspect and with slopes ranking from 10 to 30%. This area was recovered with high density coppices of resprouted individuals of *A. unedo* with heights between 1,2 and 3,0 m. The rest or plant cover are dominated mainly by shrub and grasses (to more from less abundant): *Rosmarinus officinalis*, *Quercus coccifera*, *Globularia alypum*, *Dorycnium pentaphyllum*, *Pistacia lentiscus*, and *Ulex parviflorus*; with the presence of some trees of *Q. ilex*, *Q. cerrroides* and *Juniperus oxycedrus*. In each block, we randomly chose three 50 m x 50 m plots of similar slope and aspect and one of the three treatment were applied (see above). In the center of each plot, we established a circular subplot with a 10-m radius. Within that subplot, we measured the density of *A. unedo* individuals, the number of resprouts per stump, and resprout diameter 50 cm from the base. This measurement were used because up to this height most *A. unedo* resprouts have branches and then it is difficult to measure and compare the diameter among them. The difference in *A. unedo* individuals density, resprout density and average resprout diameter between plots of each treatment was analyzed using univariate ANOVAs, with block and plot as random factors. No significant differences were found among plots for any of these variables (Table A.1). Thus, plots were similarly composed of 185 ± 9 (mean SE) strawberry individuals.

In the winter of 2006-2007, the plots within each block were assigned one of three management strategies at random. In general, these strategies included: 1) a control strategy (henceforth C), in which nothing was done to the plot (except for 20 individuals, see above); 2) a thinning treatment (henceforth T), in which resprouts were selectively removed with electric scissors (Electrocoup F3005) from all the strawberry trees on the plot (except for 20 individuals) – the 20% tallest resprouts were retained in each individual; 3) a second thinning treatment (henceforth U), in which the same selective thinning procedure was applied in

tandem with the mechanical clearing of all the surrounding understory vegetation (except for 20 individuals). However, in order to cross the treatments within each plot and thus test the intraindividual (treatment effect on individual trees) versus interindividual and interspecific effects (treatment effect on the plot), we used the following methodology. On each plot, we randomly selected 20 trees to which treatments other than the plot's assigned treatment were applied. For instance, on the C plots, 10 of the 20 trees were selectively thinned (Ct), and 10 were selectively thinned plus had vegetation cleared for 2 m around their stump (Cu). On the T plots, 10 trees were left as controls (Tc), and 10 trees were subject to selective thinning and understory clearing (Tu). Finally, on the U plots, 10 trees were left as controls (Uc), and 10 trees were subject to selective thinning (Ut). The remainder of the trees on a given plot, which received that plot's assigned treatment, are hereafter referred to as Cc, Tt, and Uu, respectively. The reduction in strawberry tree equivalent diameter (an integrative measure of resprout size obtained as the diameter of a circle with an area equivalent to the sum of the area at 50 cm of all resprouts) that resulted from treatment application is shown in Table A.2 (percentage reduction) and Table A.3. (absolute equivalent diameter values). The change in resprouts diameter average by individual (in percentage) after treatment are showed in table A.4 and the absolute values in table A.5.

Measurements of growth and induced resprouting (as described below) were performed on 50 individuals from each plot. In addition to the 20 individuals subject to cross-treatment (Ct, Cu, Tc, Tu, Uc, or Ut), another 30 individuals subject to the plot's assigned treatment (Cc, Tt, and Uu) were randomly chosen. Minimum distance between neighbouring trees with different treatment was 5 m. Overall, 600 individual strawberry trees were examined. In order to avoid edge effects, all these individuals were chosen from within a 40 m x 40 m quadrat established in the center of each plot. So as to represent the size range of strawberry trees in the area, these 50 individuals were selected so that one-third were initially large (more than 20 resprouts/stump), one-third were medium-sized (11-20 resprouts/stump), and one-third were small (up to 10 resprouts/stump).

2.4. Growth and induced resprouting measurements

We carried out the following measurements on the selected individual strawberry trees: a) retained resprout height in the winters of 2007, 2008, 2009, and 2010 (pre-treatment and 1, 2, and 3 years post-treatment); b) retained resprout diameter at 50 cm from base in the winters of 2007 and 2010 (pre-treatment and 3 years post-treatment); c) the number of dead resprouts in the winter of 2010 (3 years post-treatment). We did not measure all the resprouts found on control individuals. Instead, we measured a subset of resprouts that represented those that would have been retained had we applied selective thinning to the control trees. To characterize resprout growth, we calculated the absolute and relative increase in height of each tree's retained resprouts, both annually (2007-2008, 2008-2009 and 2009-2010) and for the entire study period (2007-2010). We also calculated the absolute and relative increase in diameter of retained resprouts for the entire study period (2007-2010). The ratio between height and diameter in 2010 was then calculated for all retained resprouts.

To quantify induced resprouting by thinning, the following measurements were taken for each individual in the winters of 2008, 2009, and 2010 (1, 2, and 3 years post-treatment): a) number of new resprouts; b) height of the most dominant new resprout; and c) overall resprout cover measured as the vertical projection of the canopy of new resprouts. We calculated, in both absolute and relative terms, the mortality of new resprouts, the increase in height of the dominant resprout, and the increase in plant cover associated with all resprouts, both annually (2008-2009 and 2009-2010) and for the entire study period (2008-2010). When more than one value for a given variable was available per tree (i.e., trees for which we measured height and diameter of different retained resprouts), we used the mean value

2.5. Data analysis

Treatment effects on retained resprouts and the new resprouts were analysed using "split-plot" ANOVAs, where plot treatment (PT) and individual treatment (IT) were fixed factors, while block and plot were random factors. The measurements

described above were the dependent variables. Relative values of dependent variables were arcsine-square-root transformed to meet homoscedastic assumptions of normal distribution. PT had three levels (C, T, and U) for both the retained resprouts and the new resprouts. IT had three levels (c, t, and u) for retained resprouts but only two (t and u) for new resprouts. We analysed the main effects of plot and individual treatments and their interaction. When the main effects were significant, we carried out post-hoc comparisons using the Tukey test. The size of live and dead control resprouts was analyzed using univariate ANOVAs; the height and diameter of the resprouts in 2009 were the dependent variables, the survival status of the resprouts (alive or dead) in 2010 was the fixed factor, and the individual tree identity was introduced as a random factor.

To examine the relationship between the growth of retained resprouts and induced resprouting, we performed a series of correlations between the growth variables (increase in height and diameter from 2007 to 2010) and the resprouting variables (reduction in the number of new resprouts, growth in height of the dominant resprout, and increase in new resprout cover from 2008 to 2010). Prior to testing these correlations, we verified that the characteristics of the new resprouts were not correlated to each other ($p > 0.05$ in all cases). As neither the mortality nor the increase in resprout cover followed a normal distribution, we used nonparametric Spearman correlations to analyze these variables. We utilized Pearson correlations for the other variables, which were normally distributed.

The percentages of relative growth were arcsine-transformed to achieve homoscedasticity. SPSS (SPSS Inc. 2006) was used for all the analyses.

3. RESULTS

3.1. Growth and mortality of retained *A. unedo* resprouts

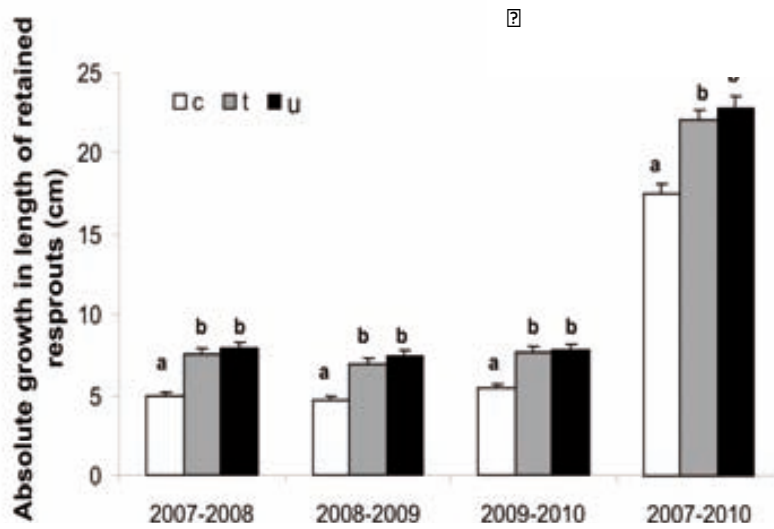
For both the entire duration and individual years of the study, the treatment applied to individual trees (IT) had a significant effect on the absolute and relative

growth in height of the retained resprouts (Table 1). The resprouts of treated individuals were longer than those of controls. There were no significant differences, however, between the two treatment types, i.e., between individuals that underwent selective thinning (t) versus selective thinning plus understory clearing (u) (Figure 2). Furthermore, three years post-treatment (in the winter of 2009-2010), the plot level treatment had a significant effect on the absolute and relative growth in height (Table 1). Absolute tree height was greater on U plots (mean±SE: 6.07±10.00) than on T (3.06±12.58cm) and C plots (2.76±7.86). Relative tree height was also greater on U plots (5.83±0.58%) than in T plots (3.68±0.25%); however, T plots showed more growth than C plots (2.81±0.17%).

Table 1. F-values, significance (p), and degrees of freedom (df) of the univariate “split-plot” ANOVA tests performed to reveal treatment effects at the plot level (PT) and the individual level (IT), as well as the impact of block (B) and plot (P) identity on the absolute and relative growth of the retained *A. unedo* resprouts. Effects on height were measured one (2007-2008), two (2008-2009), and three years (2009-2010) post-treatment, and for the entire study period (2007-2010). Effects on diameter were measured for the period 2007-2010. Values of p ($p \leq 0.05$) are in bold.

Factor	Absolute growth in height of retained resprouts										Absolute growth in diameter of retained resprouts	
	2007-2008		2008-2009		2009-2010		2007-2010		2007-2010		F	p
	df	F	P	F	P	F	p	F	p			
PT	2	0.0	0.995	0.0	0.952	4.8	0.039	1.0	0.404	0.1	0.860	
IT	2	14.6	0.000	12.0	0.000	6.2	0.002	15.6	0.000	8.8	0.000	
PT*IT	4	0.8	0.544	1.0	0.413	1.6	0.165	1.7	0.141	0.4	0.835	
B	3	0.2	0.868	1.2	0.381	2.9	0.120	1.7	0.261	0.4	0.747	
P	6	7.8	0.000	4.9	0.000	1.5	0.185	4.6	0.000	7.2	0.000	
Factor	Relative growth in height of retained resprouts										Relative growth in diameter of retained resprouts	
	2007-2008		2008-2009		2009-2010		2007-2010		2007-2010		F	p
	df	F	P	F	P	F	p	F	p			
PT	2	0.6	0.556	2.3	0.165	7.1	0.019	4.5	0.052	1.3	0.336	
IT	2	17.4	0.000	16.3	0.000	6.1	0.002	13.7	0.000	5.9	0.003	
PT*IT	4	0.7	0.590	0.4	0.828	0.3	0.911	0.4	0.814	0.3	0.898	
B	3	1.3	0.345	1.3	0.364	1.9	0.227	3.8	0.077	0.5	0.688	
P	6	3.5	0.002	2.4	0.027	3.0	0.007	2.5	0.022	4.3	0.000	

A)



B)

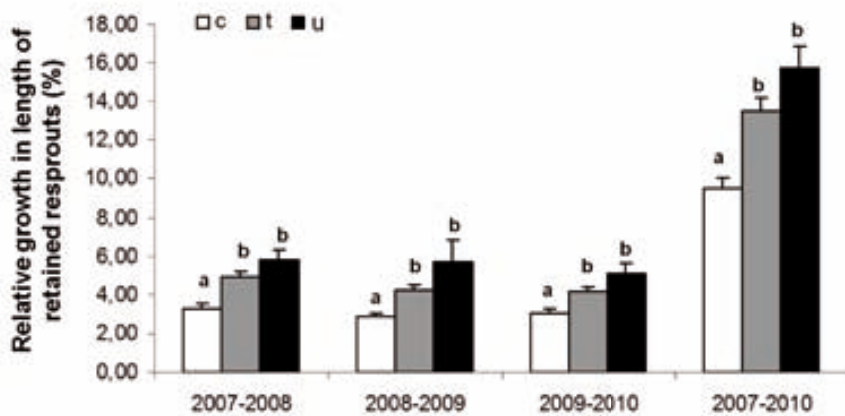


Figure 2. Mean (\pm standard error) of the absolute (A) and relative (B) growth in height of retained *A. unedo* resprouts, according to the treatment applied at the individual level: control (c) (including control trees in control plots (Cc), in selective thinning plots (Tc), and in selective thinning plus understory clearing plots (Uc)); selective thinning (t) (including selective thinning trees in control plots Ct, in selective thinning plots Tt, and in selective thinning plus understory clearing plots Ut respectively); and selective thinning plus understory clearing (u) (including selective thinning plus understory clearing trees in control plots Cu, in selective thinning plots Tu, and in selective thinning plus understory clearing plots Uu). Values of N and significant effects (p) are shown in table 1.

Resprouts also grew significantly in diameter from 2007 to 2010 in response to individual-level treatment (Table 1). Absolute and relative growth in diameter was

greater for thinned (*t* and *u*) trees (absolute growth: mean±SE: 3.66±0.17cm; relative growth: 21.57±0.78%) than for control (*c*) trees (*c* trees, absolute growth: 2.29±0.20cm; relative growth: 19.42±2.32%). There were no significant differences between treatments at the plot level and the interaction (IT*PT) was not significant (Table 1).

As for the height/diameter ratio of the retained resprouts, no significant differences were observed between the treatments, neither at the individual nor at the plot level (“Split-plot” ANOVA, $p>0.05$ for all analyses).

A vast majority of control trees had at least one dead resprout in 2010 (93.9%), a result that contrasted with the low mortality of treated trees (6.9% for *t* individuals and 14.0% for *u* individuals). The degree of self-selection may be reflected in the per capita mortality of retained resprouts: *c* trees had a much higher level of mortality per individual (34.2±1.4%) compared to treated trees (0.8±0.3% in *t* individuals and 1.2±0.3% in *u* individuals). This same pattern was found in each treatment at the plot level. For control strawberry trees (*C*), live resprouts were longer and thicker than dead ones (univariate ANOVAs, $p<0.001$, live resprouts: mean±SE 155.51±56.99 cm in height and 17.47±9.26 cm in diameter, dead resprouts: 116.31±39.66 cm in height and 9.72±5.11 cm in diameter).

3.2. Induced resprouting

In 2008, one year after treatment application, the number of new resprouts (mean±SE: 26.96±1.82 and 28.36±1.70 for *t* and *u* new resprouts, respectively) and the height of the new dominant resprout (mean±SE: 43.98±1.44cm and 49.17±1.75cm for *t* and *u* individuals, respectively) were similar for both treatment types at the individual and plot level (Table 2).

When examining induced resprouting over the course of the study (2008-2009, 2009-2010 and 2008-2010), we found no significant differences between the treatment types, at either the individual or plot level, in the mortality of new resprouts or in the growth of the new dominant resprout (Table 2). In contrast, at the individual level, treatment significantly affected new resprout cover increase

in winter 2008-2009 (Table 2) ($0.184 \pm 0.015 \text{ m}^2$ for *t* trees and $0.221 \pm 0.019 \text{ m}^2$ for *u* trees). This pattern was not observed in other years. The interaction effect was not significant (PT*IT) on measures of resprouting after treatment (Table 2). The growth in height of dominant resprout was positively correlated with the growth in height of retained resprouts, while the cover of new resprouts also showed a slight positive correlation with both the height and the diameter of the retained resprouts (Table 3).

Table 2. F and p values of the univariate “split-plot” ANOVA tests performed to test the effects of the treatment at the plot level (PT) and treatment at the individual level (IT), as well as the impact of block (B) and plot (P) identity on different estimates of *A. unedo* induced resprouting. Measurements for one (2008), two (2008-2009), and three years (2009-2010) post-treatment and the entire study period (2008-2010) are given. Values of p ($p \leq 0.05$) are in bold.

	Year	PT		IT		PT*IT		B		P	
		F	p	F	p	F	p	F	p	F	p
Number of new resprouts	2008	0.6	0,555	1.3	0,252	0.6	0,526	3.1	0,101	1.5	0,167
Height of dominant resprout	2008	0.1	0,925	3.2	0,075	0.3	0,715	0.8	0,524	5.5	0,000
Reduction in number of new resprouts	2008-2009	0.1	0,920	1.6	0,204	2.4	0,089	0.4	0,751	4.1	0,001
	2009-2010	1.0	0,410	1.5	0,223	0.1	0,886	2.8	0,126	1.8	0,107
	2008-2010	0.3	0,750	2.5	0,112	1.7	0,185	2.0	0,203	1.9	0,083
Growth in height of dominant resprout	2008-2009	1.1	0,379	0.2	0,656	1.5	0,223	0.8	0,517	1.8	0,088
	2009-2010	0.5	0,605	1.8	0,182	1.5	0,217	2.7	0,129	1.5	0,187
	2008-2010	0.4	0,654	0.2	0,639	0.9	0,393	2.7	0,135	1.5	0,175
Increase in resprout cover	2008-2009	0.7	0,501	4.3	0,038	0.9	0,417	6.5	0,020	0.8	0,535
	2009-2010	0.3	0,748	0.5	0,461	1.0	0,361	2.0	0,206	2.7	0,015
	2008-2010	0.3	0,769	3.1	0,080	1.5	0,215	3.9	0,071	2.4	0,026
df		2		1		2		3		6	

Table 3. Significance (p) and correlation coefficient (r) of the correlations between *A. unedo* retained resprout growth variables (growth in height and diameter from 2007 to 2010) and induced resprouting variables (reduction in the number of new resprouts, growth in height of dominant resprout, and increase in the cover of the new resprouts from 2008 to 2010). Significant p -values ($p \leq 0.05$) are in bold. Abbreviation: ns, not significant.

	Reduction in new resprout number			Growth in height of dominant resprout		Increase in cover	
	N	P	r- Spearman	p	r-Pearson	p	r-Spearman
Growth in height of retained resprouts	386	ns	-0.099	< 0.05	0.125	< 0.01	0.149
Growth in diameter of retained resprouts	393	ns	-0.099	ns	0.043	< 0.01	0.153

4. Discussion

4.1. Growth of *A. unedo* retained resprouts

This study demonstrates that the selective thinning of fire-generated strawberry tree coppices directly and immediately affects the resprouts of individual trees by reducing intraindividual competition, thereby allowing resprouts to grow longer and thicker (Table 1). The elimination of resprouts via thinning stimulates starch release from the lignotuber (Canadell and López-Sória, 1998), and thus promotes growth by providing the retained resprouts with more nutrients. Furthermore, thinning decreases shade, giving the resprouts that remain greater access to solar radiation and water reserves. Other studies have also shown that moderate selective thinning markedly reduces intraindividual competition and stimulates the growth of retained resprouts in the short (e.g. in *Quercus ilex*, Espelta et al., 2003; López et al., 2009; Rodríguez-Calcerrada et al., 2011) and intermediate term (e.g. in *Acacia harpophylla* (Dwyer et al., 2010) and *Q. coccinea* and *Q. velutina* (Cutter et al. 1991)).

Although selective thinning performed in tandem with understory clearing could theoretically give rise to even greater growth, we found that strawberry tree

growth was similar for both treatment types (Figure 2): the retained resprouts of treated strawberry trees grew 48% longer and 60% thicker than those of control trees. This would be the expected pattern if the increase in growth was due to a reduction of intraindividual competition (Rodríguez-Calcerrada et al., 2011) rather than a reduction of interspecific competition (i.e. competition with other shrub species present in the plot). Indeed, trees of the same species tend to share the same resources (Gustafsson and Ehrlén, 2003), especially water. The strawberry tree has a double-root system (2 to 5 m in length); its active roots located at different depths allow it to capture and use water efficiently (Clemente et al. 2005; Filella and Peñuelas, 2003). It is also, however, sensitive to drought; it reduces its photosynthetic rate when in water deficit, limiting resprout growth and delaying its phenophases (Gratani and Varone, 2003; Munné-Bosch and Peñuelas, 2004; Ogaya et al. 2003; Ogaya and Peñuelas, 2004). Castell and Terradas (1994, 1995) even reported that water is more limiting than nutrients in adult strawberry trees and their resprouts. In this context, the increased availability of water as a consequence of strawberry-tree resprout selection means that a reduction in intra-individual and intra-specific competition has a greater impact than a reduction in interspecific competition. One possible explanation for this pattern could be that understory clearing eliminates shrubs or grasses, which have shallower root systems and are therefore less likely to directly compete with *A. unedo* for water. In contrast to other studies that found that selective thinning increased growth in diameter more than growth in height for retained resprouts, as Mayor and Rodà, 1993 with *Quercus ilex*, and Lamson, 1988 with *Tilia americana*, *Acer rubrum*, *Prunus serotina*, and *Quercus rubra*, our results show that *A. unedo* resprouts grew similarly in height and diameter. Such a growth pattern contributes to the development of thicker stems that are appropriately sized for commercialization as firewood.

The self-selection process among strawberry tree resprouts is extremely slow compared to other Mediterranean resprouting tree species (Canadell et al., 1991). For many years after a fire, dead resprouts accumulate on tree stumps, giving rise to horizontal and vertical continuity in the network of dry fuel and, therefore, increasing fire risk (Pla and Rodà, 1999). Since we found that, on control trees, dead resprouts were smaller than live ones, this study shows that

selective thinning not only accelerates the self-selection process (Lamson, 1988; Rodríguez-Calcerrada et al., 2011), thus minimizing resprout mortality (only ~1% of retained resprouts on treated trees were dead, as compared to 34% of control resprouts). This clear reduction of resprout mortality improves the fire resistance of *A. unedo* coppices since it prevents the accumulation of this kind of dry and flammable fuel.

Moreover, our results demonstrate that, in the intermediate term, strawberry trees on managed plots grew more in absolute and relative height than trees on control plots (even the control trees occurring on treated plots) (Table 1). These results suggest that, after the depletion of those resources released by the stump or lignotuber, the strawberry tree also makes use of resources liberated at plot level (Robles et al., 2000). According to our second hypothesis, this pattern is a consequence of reduced intraspecific (among strawberry trees) and interspecific competition (among co-occurring species). A similar result was found by Gustafsson and Ehrlén (2003) for *Sanicula europaea*. In fact, when we applied both selective thinning and understory clearing, we observed that the absolute and relative growth of strawberry trees was greater than when we applied selective thinning alone. In contrast, we did not observe an equivalent response at the individual level; strawberry trees subject to thinning and clearing did not grow more than those that were only thinned. It therefore appears that resources were released at the plot, not the individual, level, as clearing vegetation from a 2-m radius around the base of the tree did not appear to trigger additional growth in retained resprouts. Clearing the whole plot, however, did seem to release a sufficient quantity of resources, enhancing retained resprout growth in the intermediate term. Although we do not know the range of *A. unedo*' roots, we may assume that they could easily reach > 2 m from the trunk, which is the area cleaned when the treatment was applied at individual level. If so, this would explain why there is a cleaning effect at the plot level but not at the individual level.

4.2. Effects of treatment types on induced resprouting of *A. unedo*

Management-induced resprouting may play a key role in coppice structure, especially in relation to the vertical continuity among the new resprouts and tree canopies, and thus fuel sources (Álvarez, 2011; Fernandes, 2009; Oliveras, 2009; Pla and Rodà, 1999). After thinning, the stumps of resprouting species produce a new wave of resprouts (Abbot and Loneragan, 1983; Mesléard and Lepard, 1989) that can emerge from the deeper parts of the lignotubers, up to 8 cm below ground level (Canadell et al., 1991). In this study, we show that selective thinning did indeed trigger the appearance of new resprouts. However, and contradicting our third hypothesis, this effect was not impacted by the clearing of vegetation (Table 2). As with the growth of the retained resprouts, the vigour of resprouting appears to stem from a reduction in intraindividual competition resulting from selective thinning, rather than a reduction in interspecific competition produced by clearing. Our findings suggest, therefore, that if there is less competition between the resprouts of a single tree, an increase in resprouts growth will be induced. Understory clearing, a common fire-prevention practice in Mediterranean woodlands, does not enhance induced resprouting and therefore does not contribute to vertical fuel continuity. However, two years after the treatment, the cover increase of new resprouts on thinned and cleared plots was greater than on thinned plots. This result might reflect that the reduction of competition for sunlight, as shade cast by other shrub species is reduced, promotes the lateral spread of resprouts. These differences were not observed later in the study, probably because cleared plots recovered their shrub layer, restoring competition for solar radiation.

In contrast to studies of other species such as *Q. ilex* (López et al., 2009; Rodríguez-Calcerrada et al., 2011) and contrary to our fourth hypothesis, we did not find a negative relationship between the growth of retained resprouts and the vigour of induced resprouting. We even found significant, slightly positive correlations between these variables (Table 3). We also did not find any relationship between the mortality rate of new resprouts and the growth of retained resprouts. Our hypothesis was based on the idea that the new *Q. ilex* resprouts consume carbon and water from the stump and thus compete with their

fellow resprouts (López et al., 2009). Surprisingly, this pattern was not seen for the strawberry tree in our study area; the two processes proved to be relatively independent. One explanation may be that the competition between new resprouts and retained resprouts was not strong enough to impact retained resprout growth, perhaps because selective thinning was very intensive, and the release of resources equally sustained retained and new resprouts. The resources used for induced resprouting, at least in the initial phases, might mainly come from the lignotuber and are thus less accessible to retained resprouts, which rely primarily on the root system (Canadell and López-Soria 1998).

5. Conclusions

Given that burned Catalanian forests are being increasingly dominated by strawberry-tree coppices (L. Quevedo, unpublished results), it is important to develop post-fire management strategies. We conclude that the selective thinning of strawberry trees, whether or not it is accompanied by understory clearing, has a positive effect on the absolute and relative growth of retained resprouts. This positive effect is observed both at the individual (short-term effect) and plot level (intermediate-term effect). Moreover, the appearance of new, induced resprouts, which limits the effectiveness of selective thinning in structuring coppices, does not seem to significantly affect retained resprouts, at least in the first few years. Indeed, the greater growth of retained strawberry-tree resprouts subsequent to thinning improves the vertical structure of *A. unedo* coppices. Resprouts also grew thicker, a type of growth that is crucial if they are eventually to be used commercially as firewood. Dead fuel was no longer present at tree stumps, as the thinning promoted the resprouts' self-selection process and thus enhanced fire prevention (Moreira et al., 2011).

The results obtained for the combined strategy of selective thinning and understory clearing are also relevant to the management of Mediterranean forests. Clearing is not a treatment that is commonly applied to large areas of woodlands, but it is an important part of the maintenance of both protective anti-

fire strips around traffic routes and green firewalls, as well the urban-forest interface (Bradley, 1984; Robles et al., 2000). In the mid term, clearing increases the growth of retained resprouts without increasing growth of new resprouts and, therefore, does not contribute to vertical fuel continuity.

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Appendices

Table A.1. F-values and p-values of the univariate ANOVA tests performed to determine if the given characteristics of *A. unedo* coppices were similar prior to treatment (control plots, C; plots with selective thinning, T; and plots with selective thinning plus understory clearing, U). Significant F-values ($p \leq 0.05$) are in bold. Mean and standard error are indicated for each treatment.

	Block		Plot		Mean \pm SE and rank		
	F	<i>p</i>	F	<i>p</i>	C	T	U
Individuals/ha	8.0	0.009	0.1	0.882	1265 \pm 625	1082 \pm 498	1122 \pm 487
Resprouts/ha	2.8	0.107	0.5	0.593	11730 \pm 2740	14340 \pm 4677	15494 \pm 7154
Equivalent diameter (cm)	1.0	0.441	0.5	0.624	5.13 \pm 1.51	6.46 \pm 0.45	5.99 \pm 0.48

Table A.2. Percentage reduction (mean \pm SE) in the equivalent diameter (cm) of individual strawberry trees following treatment application in the winter of 2007.

Plot Treatment (PT)	Individual treatment (IT)	Trees (PT*IT)	N	Block 1	Block 2	Block 3	Block 4
	c	Cc	30	-	-	-	-
Control (C)	t	Ct	10	24.51 \pm 11.42	29.17 \pm 4.42	31.34 \pm 0.00	31.57 \pm 2.77
	u	Cu	10	32.56 \pm 10.70	28.86 \pm 2.46	31.56 \pm 3.11	34.89 \pm 3.67
	c	Tc	10	-	-	-	-
Selective thinning (T)	t	Tt	30	32.76 \pm 11.80	34.20 \pm 1.84	35.21 \pm 2.03	29.44 \pm 2.21
	u	Tu	10	28.72 \pm 11.13	36.28 \pm 4.97	32.80 \pm 3.16	30.68 \pm 2.58
	c	Uc	10	-	-	-	-
Selective thinning plus understory clearing (U)	t	Ut	10	30.89 \pm 2.71	37.63 \pm 2.68	34.78 \pm 3.06	31.53 \pm 3.21
	u	Uu	30	27.45 \pm 2.31	37.53 \pm 2.27	36.45 \pm 2.42	34.80 \pm 2.02

Note: Control plots (C) included control trees (Cc), trees subjected to selective thinning (Ct), and trees subjected to selective thinning plus understory clearing (within 2-m radius of tree base) (Cu); selective thinning plots (T) included control trees (Tc), trees subjected to selective thinning (Tt), and trees subjected to selective thinning plus understory clearing (Tu); selective thinning plus understory clearing plots (U) included control trees (Uc), trees subjected to selective thinning (Ut), and trees subjected to selective thinning plus understory clearing (Uu).

Table A.3. Absolute equivalent diameters (mean±SE) (cm) of individual strawberry trees following treatment application in the winter of 2007.

Plot Treatment (PT)	Individual treatment (IT)	Trees (PT*IT)	N	Block 1	Block 2	Block 3	Block 4
Control (C)	c	Cc	30	5.95 ± 3.45	7.74 ± 0.61	5.71 ± 0.49	5.17 ± 0.51
	t	Ct	10	3.90 ± 2.41	5.32 ± 0.79	4.03 ± 0.49	4.25 ± 0.75
	u	Cu	10	3.22 ± 2.15	5.96 ± 0.79	3.83 ± 0.26	3.82 ± 0.71
Selective thinning (T)	c	Tc	10	4.81 ± 2.64	6.05 ± 2.27	4.58 ± 1.20	4.70 ± 0.82
	t	Tt	30	3.92 ± 1.95	4.25 ± 0.41	3.87 ± 0.34	3.71 ± 0.27
	u	Tu	10	4.66 ± 2.55	4.91 ± 0.84	2.92 ± 0.56	3.71 ± 0.60
Selective thinning plus understory clearing (U)	c	Uc	10	5.03 ± 0.87	4.05 ± 0.68	6.30 ± 1.19	6.24 ± 0.90
	t	Ut	10	3.92 ± 0.41	2.62 ± 0.50	3.71 ± 0.77	3.96 ± 0.32
	u	Uu	30	3.97 ± 0.37	2.93 ± 0.22	3.37 ± 0.27	4.76 ± 0.54

Note: For abbreviations, see table A. 2.

Table A.4. Increase (%) of resprout diameter average of each individual of strawberry trees (mean±SE) following treatment application in the winter of 2007.

Plot Treatment (PT)	Individual treatment (IT)	Trees (TP x TI)	N	Block 1	Block 2	Block 3	Block 4
Control (C)	c	Cc	30	-	-	-	-
	t	Ct	10	58.98	67.63	80.48	62.39
	u	Cu	10	49.53	51.79	68.75	50.40
Selective thinning (T)	c	Tc	10	-	-	-	-
	t	Tt	30	56.24	70.06	60.40	52.96
	u	Tu	10	66.46	65.41	51.78	59.46
Selective thinning plus understory clearing (U)	c	Uc	10	-	-	-	-
	t	Ut	10	56.58	66.90	62.62	52.28
	u	Uu	30	72.03	61.82	52.13	59.96


Note: For abbreviations, see table A. 2.

Table A.5. Resprout diameter (cm) average of each individual (mean±SE) following treatment application in the winter of 2007.

Plot Treatment (PT)	Individual treatment (IT)	Trees (TP x TI)	N	Block 1	Block 2	Block 3	Block 4
	c	Cc	30	1.60 ± 0.044	1.92 ± 0.045	1.51 ± 0.042	1.33 ± 0.034
Control (C)	t	Ct	10	2.08 ± 0.155	3.08 ± 0.126	2.24 ± 0.111	2.42 ± 0.128
	u	Cu	10	2.23 ± 0.122	3.05 ± 0.160	1.92 ± 0.128	2.23 ± 0.121
	c	Tc	10	0.99 ± 0.047	1.69 ± 0.098	1.26 ± 0.070	1.25 ± 0.067
Selective thinning (T)	t	Tt	30	2.15 ± 0.060	2.62 ± 0.089	2.24 ± 0.071	2.19 ± 0.077
	u	Tu	10	2.45 ± 0.098	2.62 ± 0.136	1.84 ± 0.142	2.01 ± 0.128
Selective thinning plus understory clearing (U)	c	Uc	10	1.18 ± 0.062	1.08 ± 0.046	1.39 ± 0.064	1.81 ± 0.084
	t	Ut	10	1.88 ± 0.083	1.69 ± 0.121	2.18 ± 0.126	2.30 ± 0.089
	u	Uu	30	2.03 ± 0.069	1.83 ± 0.058	1.93 ± 0.067	2.87 ± 0.104

Note: For abbreviations, see table A. 2.

CHAPTER 3

Post-fire forestry management improves fruit quality in forest coppices dominated by *Arbutus unedo* L. 

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(en revisió)



Post-fire forestry management improves fruit quality in forest coppices dominated by *Arbutus unedo* L.

Abstract/ Resum

In Mediterranean ecosystems, post-fire forestry management practices are often used to improve forest structure and to reduce the risk of fire in coppices of resprouting species. Such practices enhance tree growth (i.e., height), probably because they release resources. On the one hand, resource release may stimulate reproduction. On the other hand, tree species that are regenerating after a fire may already face a delay in reproduction, and this delay may be lengthened if species mainly invest these additional resources in growth. Within this theoretical framework, it is poorly understood how different forest management practices affect the reproductive abilities of forest species. In this study, we analyzed the effect of two post-fire forestry treatments (selective thinning of resprouts and selective thinning of resprouts plus understory clearing) on a Mediterranean coppice dominated by the resprouter species *Arbutus unedo* L.; in particular, we examined how the treatments affected this species' reproductive ability (flower and fruit production at the tree and stand level, as well as fruit quality). Our results show that the treatments had no effects on the number of flowers and mature fruits per individual. Meanwhile, mature fruit dry mass and seed set were greater in plots that had been both thinned and cleared than in control plots and plots that had only been thinned. This pattern was reversed for seed abortion rate: it was lower in plots that had been thinned and cleared. The dry mass of developed seeds did not differ among treatments. At the stand level, the percentage of strawberry trees that flowered, the percentage of strawberry trees that bore fruit, the dry biomass of mature fruits per hectare, and the number of developed seeds per hectare were not affected by these treatments. Other studies have shown that these two forest management practices can improve the vertical and horizontal structure of *A. unedo* coppices that are regenerating post fire; this study demonstrates that such practices do not negatively affect the species' reproductive success and, in fact, might even enhance it. Consequently, these forestry practices might ensure the natural

regeneration of populations of this species, as well as the availability of food for local fauna. It is thus highly recommended that such practices be used to manage coppices dominated by resprouter species following fire, especially in situations where the growth of the forest canopy has stagnated and/or reproduction of forest species has been delayed.

Keywords: selective thinning; understory cleaning; strawberry tree; fruit production; flower production; Mediterranean

En els ecosistemes mediterranis, les pràctiques de gestió forestal postincendi s'utilitzen sovint per millorar l'estructura forestal i per reduir el risc d'incendi en bosquines d'espècies rebrotadores. Aquestes pràctiques milloren el creixement de l'arbre (per exemple, l'alçada), probablement degut a que alliberen recursos. Per una banda, l'alliberament de recursos pot estimular la reproducció. Per altra, les espècies arbòries que s'estan recuperant després d'un foc poden patir un retard en la reproducció, i aquest retard pot allargar-se si l'espècie inverteix aquests recursos sobretot en el creixement. Dins d'aquest marc teòric, hi ha pocs treballs que analitzin com diferents pràctiques de gestió forestal afecten les habilitats reproductives de les espècies forestals. En aquest estudi, analitzem l'efecte de dos tractaments forestals postincendi (selecció de rebrots i selecció de rebrots acompanyada de la desbrossada del sotabosc) en una bosquina mediterrània dominada per l'espècie rebrotadora *Arbutus unedo* L.; en concret, vam analitzar com els tractaments van afectar l'habilitat reproductiva d'aquesta espècie (producció de flors i fruits a nivell individual i de parcel·la, així com la qualitat dels fruits). Els nostres resultats mostren que els tractaments no van tenir efectes en número de flors i de fruits madurs per individu. Mentrestant, el pes sec dels fruits madurs i el número de llavors desenvolupades per fruit va ser superior a les parcel·les en què s'havia aplicat la selecció de rebrots i la desbrossada del sotabosc, respecte les parcel·les control i les parcel·les que només se li havia aplicat la selecció de rebrots. Aquest patró va ser invertit per la taxa d'avortament de llavors: aquesta va ser inferior a les parcel·les que havien estat seleccionades i desbrossades. El pes sec de les llavors desenvolupades no va variar entre tractaments. A nivell de parcel·la, el percentatge d'arboços que van florir, el percentatge d'arboços que van fructificar, el pes sec de la biomassa

de fruits madurs per hectàrea, i el número de llavors desenvolupades per hectàrea no van ser afectats per aquests tractaments. Altres estudis han mostrat que aquestes dues pràctiques poden millorar l'estructura vertical i horitzontal de les bosquines d'*A. unedo* que estan regenerant després del foc; aquest estudi demostra que aquestes pràctiques no afecten negativament l'èxit reproductiu de l'espècie i, de fet, fins i tot podria millorar-lo. Conseqüentment, aquestes pràctiques forestals podrien assegurar la regeneració natural de les poblacions d'aquesta espècie, així com la disponibilitat de menjar per a la fauna local. És per això que recomanem molt que aquestes pràctiques siguin utilitzades per gestionar bosquines dominades per espècies rebrotadores després del foc, especialment en situacions on el creixement de la massa forestal s'hagi estancat i/o la reproducció de l'espècie forestal s'hagi retardat.

1. Introduction

Forest ecosystems around the world experience natural and anthropogenic disturbances (Bengtsson et al., 2000). Such disturbances change forest structure and alter species composition (Dellasala et al., 2006). For example, wild forest fires or forestry practices related to wood harvesting may partially or completely remove the forest canopy (Rodrigo et al. 2004; Arnan et al. 2009) and cause changes in the composition of the associated plant species community (Trabaud, 1994; Arnan et al. 2007; Puerta-Piñero et al. 2012). However, even when species are able to recover, disturbance can simplify forest structure (Piussi, 1992; Hutto and Gallo, 2006; Herrando et al., 2009; Moreira et al., 2011) and delay species reproduction for several years (Verkaik and Espelta, 2006; Rodríguez-Calcerrada et al., 2011).

Forest fires occur naturally in Mediterranean ecosystems, and fire regimes play a central role in shaping Mediterranean landscapes (Gill et al., 1981; Piuksi, 1992; Naveh, 1994; Trabaud, 1994; Rodrigo et al., 2004; Moreira et al., 2011). Mediterranean ecosystems have also long been subject to human influences (Hobbs et al., 1995), mainly those associated with forest-related activities. Many dominant tree and/or shrub species found in Mediterranean basin communities tend to recover quickly after fire (Pausas, 2001; Rodrigo et al., 2004) or clearing (Ducrey and Turrel, 1992; Pelc et al., 2011; Quevedo et al., 2013) since they have, in general, strong resprouting abilities (Pausas et al., 1999; Arnan et al., 2007; Paula et al., 2009; Quevedo et al., 2007) or can even germinate from soil or canopy seed banks (Tapias et al., 2004; Rodrigo et al., 2012). As a result of their highly efficient regeneration mechanisms, these dominant species usually produce a large number of seedlings per hectare, in the case of seeder species, or of resprouts per hectare, in the case of resprouter species. Consequently, growth of the forest canopy stagnates, and recovery of the canopy's vertical structure is hindered (Espelta et al., 2003). Moreover, even when dominant species do not efficiently recover following disturbances, resprouter species in the understory may become dominant (Rodrigo et al., 2004; Arnan et al., 2013); they can also produce a large number of resprouts per individual and, consequently, demonstrate great horizontal and vertical continuity. Regardless of the exact situation, the reproductive capacities of certain forest species are negatively affected; for instance, woody species need to reach sexual maturity, or a minimum size, to produce flowers and fruits following disturbances (Obeso, 2002; Verkaik and Espelta, 2006; Moya, 2008). Moreover, the high density of stems and resprouts results in a high level of competition for resources, such as water, light, and nutrients, and consequently, reproduction may be delayed even further (Espelta et al., 2003).

However, it is crucial that forest species recover the ability to reproduce following disturbances since sexual reproduction is the mechanism by which genetic variability is reincorporated into populations (Sánchez-Humanes and Espelta, 2011), e.g., through the establishment of new seedlings within a community or colonization of new areas through seed or fruit dispersal (Herrera, 1982; Debussche and Isenmann, 1989; Aparicio et al. 2008; Rost et al., 2012a).

Furthermore, the flowers and fruits produced after disturbances serve as important sources of food for forest fauna, such as birds, mammals, and insects (Hulme, 1997; Arnan et al., 2007; Jordano et al., 2007; Christianini and Oliveira, 2010; Virgós et al., 2010; Arnan et al., 2011) and thus help these animal populations recover (Kollmann, 2000; Arnan et al., 2006; Virgós et al., 2010; Rost et al. 2012b).

When a regenerating forest has a high seedling density and/or a large number of resprouts per tree, forestry management practices are often used to promote stem and resprout growth to improve forest structure (Ducrey and Toth, 1992; Gracia et al., 1999; Espelta et al. 2003; González-Ochoa et al., 2004; De las Heras et al. 2012) and to reduce the risk of fire (Cañellas et al., 2004). It has been demonstrated that selectively thinning resprouts in areas dominated by resprouter species (e.g., holm oaks, oaks, and strawberry trees) stimulates the growth (height and diameter) of the remaining resprouts (Retana et al., 1992; Espelta et al., 2003; Rodríguez-Calcerrada, et al. 2008; Quevedo et al., 2013), probably due to the release of resources. The resources released by the plants are then allocated to the most important physiological processes, namely vegetative growth and reproduction (Levins, 1968; Obeso, 2002; Sánchez-Humanes et al., 2011); consequently, such forest management practices may even stimulate reproduction (Healy et al., 1999; González-Ochoa et al., 2004; Verkaik and Espelta, 2006). However, the relationship between forest management practices and the reproductive abilities of forest species remains poorly explored.

In this study, we analyzed whether applying post-fire forestry treatments to a Mediterranean forest dominated by the resprouter species *Arbutus unedo* L., or the strawberry tree, modified this species' reproductive ability. Our goal was to assess whether selectively thinning resprouts and mechanically clearing understory vegetation in strawberry tree coppices affected flower and fruit production at the individual (tree) and plot (stand) level; we also examined changes in fruit quality. In particular, we investigated whether these treatments resulted in the following changes: i) an increase in flower and fruit production at

the individual level; ii) an improvement in fruit and seed quality (as measured by mass); and iii) an increase in fruit production at the stand level.

2. Materials and methods

2.1. Study area

This study was carried out in the northern part of the Baix Llobregat region (41° 35' N, 1° 52' E, Catalonia, north eastern Iberian Peninsula), in a 238-hectare area previously burned by three wildfires (in 1985, 1986, and 1994) (Quevedo et al., 2013). Prior to the fires, this area was covered by forests dominated by *Pinus halepensis* (Mapa de Cultivos y Aprovechamientos, 1980); after these recurrent fires, the forest that developed became largely dominated by strawberry trees producing a large number of resprouts per individual (12 ± 0 , mean \pm SE). These coppices have a high degree of vertical and horizontal continuity and require special management. The study area is located 390 to 500 m above sea level and occurs within the strawberry tree's natural range in Catalonia. It is characterized by a dry Mediterranean climate (according to the Thornthwaite moisture index); average annual temperature is 13.5 °C, and mean annual precipitation ranges from 650 to 700 mm.

2.2. Study species

The strawberry tree (*Ericacea*) is a tree species that displays a low degree of sclerophylly (Gratani and Ghia, 2002a). It naturally occurs in North Africa, the Middle East, Macaronesia, and western and southern Europe; it is also widely distributed across the Iberian Peninsula and the Balearic Islands. It resprouts vigorously after disturbances (such as clearing or fire) and has a slow growth rate. Although it usually occurs as a companion species in Mediterranean forests, it can develop its own wooded coppices after recurrent fires (Arnan et al., 2013).

It blooms in the fall and has hanging panicles—with 40–50 white flowers each (Sealy and Webb, 1950)—that are pollinated by relatively large bees, such as

Apis mellifera and bumble bees in the genus *Bombus* (Herrera, 2004). It produces round red fruits of 7 to 30 mm in diameter, which can contain up to 40 seeds (Sealy and Webb, 1950; Moro, 1995). Preliminary observations have suggested that these fruits take a year to mature, and as a result, flowers and fruits are present at the same time on the same tree in the fall and winter (Galán et al., 1998).

The strawberry tree demonstrates a certain amount of interannual variation in fruit production (Herrera, 1998), although the species does not engage in masting. It does also exhibit large interindividual variation in fruit production. A single individual can produce between 2.6 and 10 kg of fruit per year (Gomes and Canhoto, 2009; Molina et al., 2011), and its fruits are a source of food for animals, especially birds and mammals (Arianoutsou and Diamantopoulos, 1985).

2.3. Experimental design and post-fire forestry treatments

In 2006, we established four sampling blocks in the study area; the blocks were located at sites with high densities of regenerating strawberry trees and that were also quite homogeneous in terms of slope (between 10 and 30%) and orientation (southwest). Within each block, we established three square 0.25-ha plots (50 m x 50 m) at randomly chosen locations. All the plots had similar strawberry tree and resprout densities (Quevedo et al., 2013).

In the winter of 2006-2007, one of three types of experimental treatments were randomly assigned and applied to the plots within each block: 1) a control treatment, where nothing was done (hereafter C); 2) a treatment in which resprouts were selectively thinned (hereafter T) using electric shears (Electrocoup F3005) and a handsaw (20% of each strawberry tree's largest-diameter resprouts were retained); and 3) a combined treatment in which resprouts were selectively thinned as just described and understory vegetation in the strawberry tree coppices was mechanically cleared (hereafter U) (Quevedo et al., 2013).

2.4. Measures of reproduction

To observe the changes that occur during the strawberry tree's annual phenological cycle and to choose which reproductive variables to measure, we followed the species' reproductive phenology for a year (June 2006 to June 2007) before applying the treatments. This approach allowed us to more accurately define the following different phenophases that we evaluated in this study: a) flower production: we focused on fully formed white flowers and excluded wilted and brown flowers, which resulted from cold or rainy conditions; b) green fruit production: we focused on all small hard developing fruits that showed no signs of maturity; and c) mature fruit production: we focused on all the red and soft fruits that had already reached maturity.

To evaluate the effects of the post-fire forestry treatments on strawberry tree reproduction at the individual level, we selected 15 individual strawberry trees per plot and counted the number of flowers and fruits (green and mature) they produced. All the trees were located within a 40-m x 40-m area in the plot's center in order to minimize edge effects; consequently, any individuals located in the remaining 5-m strip along the plot's border were excluded. Individuals were selected based on their original number of resprouts (i.e., before thinning on the T and U plots); one-third had many resprouts (more than 20 per individual), one-third had intermediate numbers of resprouts (between 11 and 20 per individual), and one-third had low numbers of resprouts (no more than 10 per individual). Our aim was to sample equal numbers of strawberry trees with different resprout numbers. Flowers and fruits were counted over a 32-month period, first on a weekly basis (from June 2007 to January 2008) and then on a biweekly basis (from February 2008 to December 2009). In this way, we managed to follow two complete cohorts, or generations (2007-2008 and 2008-2009), following treatment application; we defined our cohorts as beginning when flowers were produced during a given year and ending when those flowers produced mature fruits the following year.

2.5. Fruit quality

To study the effects of the post-fire forestry treatments on strawberry tree fruit quality, 150 mature fruits (50 from each resprout-number category) were collected at random from each plot in 2008 and 2009 (for the first and second cohort, respectively). The fruits were taken from different trees than those whose flowers and fruits had been counted; between 1 and 10 fruits were collected from each tree (1,800 per cohort). Fruits were randomly sampled from along the entire surface of the tree crown. All the fruits were dried in an oven at 70 °C for a minimum of 3 days; each fruit's dry mass (in mg) was measured using a PRECISA 125A (PACISA) analytical balance (± 0.1 mg).

The fruits were then soaked in a 3% hydrogen peroxide solution for at least 15 days, so as to soften them and make it easier to extract their seeds. We developed this method based on the results of previous tests because there is no documented procedure for rehydrating this type of fruit. The seeds were extracted by hand; developed seeds were separated from undeveloped seeds on the basis of embryo presence. The undeveloped seeds clearly contained nothing below their surface casings. The developed seeds were then dried in the oven at 60 °C for 24 hours. Seeds from each fruit were then weighed together (± 0.01 mg) using an analytical balance.

2.6. Presence of caterpillars

During this study, strawberry trees on our plots were affected by an infestation of the caterpillar *Euproctis chrysorrhoea* L. This univoltine caterpillar (*Lymantriidae*) tends to reach extreme densities and the young larvae, when in embryonic diapause, feed on the leaves of the strawberry tree (Frago et al., 2010). As a result, such infestations can negatively affect strawberry tree reproduction (Soria and Notario, 1990; Hódar et al., 2003). Therefore, in addition to quantifying strawberry tree reproduction, we also counted the number of caterpillar bags that were present on each tree. A caterpillar bag is all the filamentous structures that are grouped in sets on tree leaves; it resembles a closed bag and contains *Euproctis chrysorrhoea* L. eggs, larvae, or caterpillars.

To analyze whether infestation intensity differed among treatments, we first calculated the two following variables: a) the percentage of infested trees found on each plot; and b) the degree of infestation, i.e., the mean number of bags found on each strawberry tree (excluding trees without bags). Both variables were calculated for data obtained in 2007, 2008, and 2009. We then explored the effects of the treatments on these variables using general linear mixed models (function *lme* in package *nlme*, R v. 2.13.0; R Development Core Team, 2008). The degree of infestation was square-root transformed in order to achieve homoscedasticity. Treatment and year were fixed factors; block ID and plot ID, as well as individual tree ID in the case of the degree of infestation, were random factors. We then obtained the ANOVA tables summarizing the model results, which provided global p-values for the fixed factors. Unlike in our analyses of reproduction, we included data from 2007 in these analyses to determine if the caterpillars had different effects on the first (2007-2008) versus the second (2008-2009) study cohort.

There were significant differences among years in the percentage of infested trees found on the plots. Percentage infestation was higher in 2009 than in 2008 and 2007 ($F_2 = 19.5$, $p < 0.0001$; mean \pm SE: $6 \pm 1\%$, $40 \pm 18\%$, and $64 \pm 23\%$ for the years 2007, 2008, and 2009, respectively), but there were no significant differences among the treatments nor was the interaction between year and treatment significant ($p > 0.05$).

The degree of infestation differed significantly among years ($F_2 = 85.6$, $p < 0.0001$), and the treatment-by-year interaction was significant as well ($F_4 = 9.0$, $p < 0.0001$). Thus, globally, there were more caterpillar bags per tree in 2009 than in 2008. Furthermore, in 2009, there were more caterpillar bags on trees on U plots than on trees on C and T plots; this difference was not present in 2007 and 2008 (Appendix 1). These results mean that it was hard to untangle the effects caused by the caterpillars from those caused by the treatments for 2009. For this reason, the analyses focusing on strawberry tree reproduction were conducted separately for each cohort: in the first cohort, the degree of infestation was similar among treatments, but in the second cohort, the degree of infestation differed among treatments.

2.7. Data analysis

To assess the effects of the treatments on strawberry tree reproduction, we examined the three categories of response variables: a) individual-level production; b) fruit quality; and c) stand-level production. Each category included the following variables:

- 1) Individual-level production: a) number of flowers per individual; b) number of mature fruits per individual; c) flower success rate: percentage of flowers per individual that produced green fruits (analogous to fruit set in most other studies); d) fruit maturation rate: percentage of mature fruits resulting from green fruits produced by an individual the same year. Unproductive individuals (in terms of flowers and/or fruits) were excluded from the analysis.
- 2) Fruit quality: e) dry mass of mature fruits (g); f) seed production per fruit: number of developed seeds in each fruit (analogous to seed set in most other studies); g) seed abortion rate: percentage of undeveloped seeds relative to the total number of seeds in a fruit; h) dry mass of developed seeds (mg).
- 3) Stand-level production: i) percentage of individual trees bearing fruits per plot; j) dry biomass of mature fruits per hectare (kg): calculated for each plot as the sum of the production in kg for each of the three resprout-number categories—production in each category was determined by multiplying the number of trees in the category by the percentage of trees bearing fruits by the mean per-tree fruit production in the category by the dry mass of each fruit; k) number of developed seeds per hectare: calculated for each plot by multiplying the average number of mature fruits by per-fruit seed production by the percentage of trees bearing fruits by the number of individuals per hectare.

To achieve homoscedasticity, the variables were subject to either square-root, logarithmic, or arc-sine square-root transformations, depending on the variable. General linear mixed models were then used to analyse the effects of the treatments on these variables. Treatment was a fixed factor in all the models.

The random factors differed depending on the response variable category being examined: block ID and plot ID were random factors in the individual-level-production models; block ID, plot ID, and individual ID were random factors in the fruit-quality models; and block ID was the only random factor in the stand-level-production models.

Because plants have limited amounts of resources (i.e., water and nutrients), they must decide how to allocate these resources to vegetative growth versus reproduction (principle of allocation; Levins, 1968). To assess how the strawberry tree allocates its resources, we tested whether there was a negative correlation between the number of mature fruits produced by a tree and the relative growth (mean increase in height) of its resprouts (growth-reproduction trade-off) using data from 2008. For each tree, the number of mature fruits was divided by the number of resprouts present at the time of measurement, to make the measurements comparable to relative growth. On the control plots, the relative growth of the resprouts (change in mean height) was estimated using all the resprouts present on sampled individuals (not just those that had been considered to be retained); on the T and U plots, relative growth was estimated using the retained sprouts. This analysis only included data from 2008 because, in 2009, the *Euproctis chrysorrhoea* L. infestation had spread and it was thus difficult to exclude its effects on strawberry tree reproductive phenology.

To achieve homoscedasticity, the number of mature fruits was square-root transformed. The growth-reproduction trade-off was analyzed using a general linear mixed model. The number of mature fruits was the response variable, treatment and relative sprout growth were fixed factors, and the tree's initial number of resprouts (i.e., before treatment application) was a covariable. Block ID and plot ID were included as random factors.

3. Results

3.1. Description of *Arbutus unedo* L. reproductive phenology

We were able to clearly establish the phenophases that make up the strawberry tree's reproductive cycle (Figure 1), which displays a certain complexity: since the species' fruits take a year to mature, mature fruits and flowers co-occur in the canopy in the fall and winter. Flower production peaked in mid-November. Green fruit production peaked between late March and mid-May, and total fruit volume continued to increase throughout the summer as the fruits started to mature (changing in colour from green to yellow-orange). Fruit maturation reached its peak during the second half of October. This temporal pattern did not vary among treatments or between cohorts (Figure 1), although both flower production and fruit production were lower the second year.

3.2. Individual-level production

Neither the number of flowers nor the number of mature fruits per individual differed significantly among treatments for either of the two cohorts (Table 1). For the first and second cohort, the mean number of flowers per individual was 360 ± 62 (mean \pm SE) and 446 ± 58 , respectively, and the mean number of mature fruits per individual was 48 ± 7 and 17 ± 4 , respectively. Similarly, neither flower success rate nor fruit maturation rate differed significantly among treatments for either cohort (Table 1; flower success rate— $32 \pm 2\%$ and $18 \pm 2\%$ —and fruit maturation rate— $28 \pm 2\%$ and $22 \pm 2\%$ —for the first and second cohorts, respectively).

The number of mature fruits per resprout differed among treatments ($F_2 = 9.9$, $p = 0.013$); it was higher on the T and U plots than on C plots, but there was no difference between the T and U plots. There was no relationship between the number of mature fruits/resprout and relative resprout growth ($F_1 = 3.5$, $p = 0.06$), nor was there a treatment-by-growth interaction ($F_2 = 1.3$, $p = 0.27$). This result suggests that there was no trade-off between reproduction and growth.

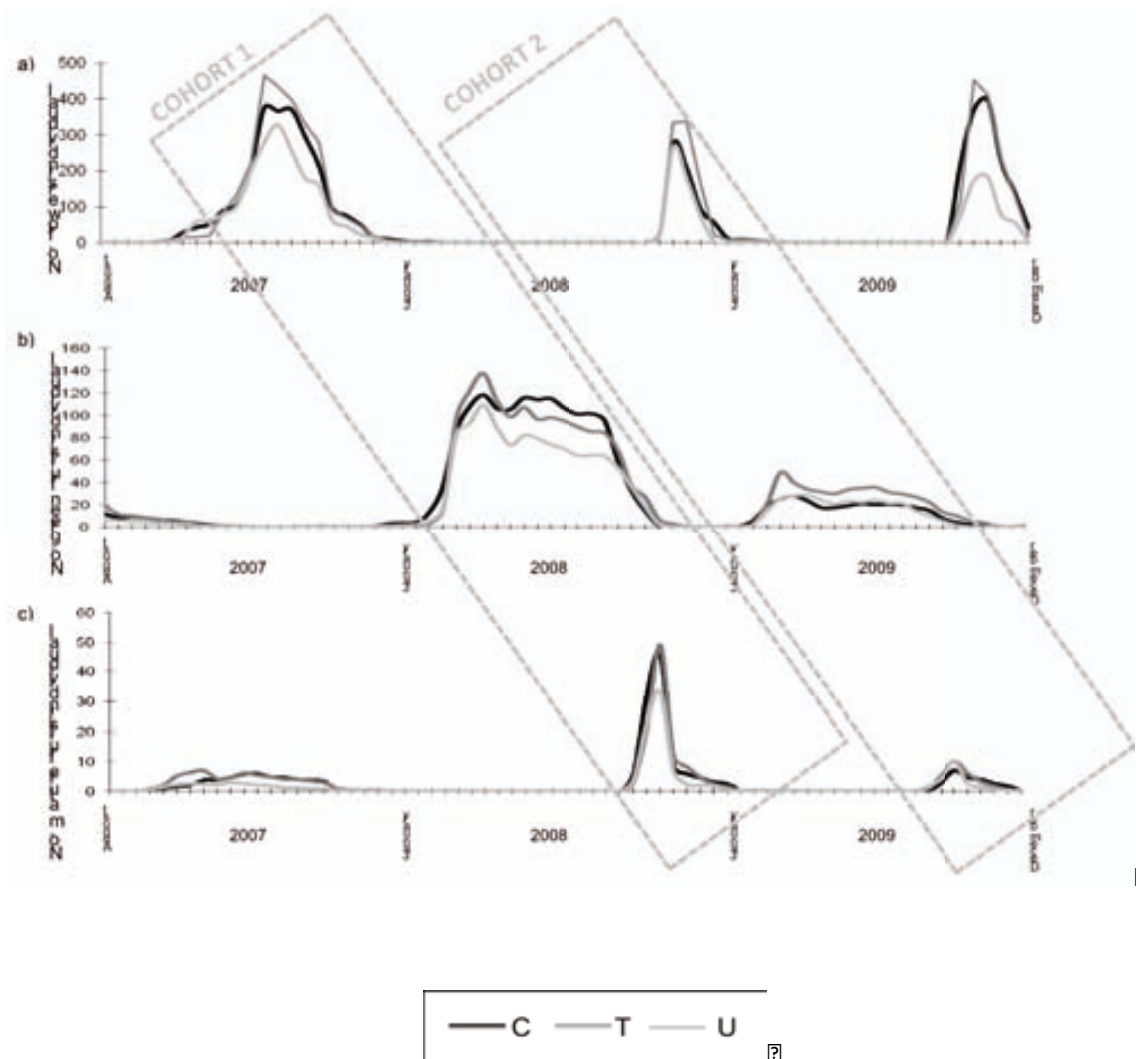


Figure 1. Reproductive phenology of *Arbutus unedo* L. in the study plots (2007-2009) as represented by three variables: a) number of flowers/individual, b) number of green fruits/individual, and c) number of mature fruits/individual. For each date, the means for individuals on control plots (C, black), plots that experienced selective thinning (T, dark gray), and plots that experienced selective thinning+understory clearing (U, light gray) are provided. Flower, green fruit, and mature fruit production peaks are represented on the dates associated with the maxima on the graphs; mature fruits were harvested during the period of peak production.

3.3. Fruit quality

Mature fruit dry mass and seed production were higher on the U plots than on the C and T plots for the first cohort, but there was no difference among treatments

for the second cohort (Table 1, Figure 2). In contrast, the seed abortion rate was lower on the U plots than on the C and T plots for the second cohort, while there was no such difference among treatments for the first cohort (Table 1, Figure 2). The dry mass of developed seeds did not differ among treatments for either of the two study cohorts (Table 1); mean (\pm SE) dry mass per seed was 1.16 ± 0.01 mg and 1.14 ± 0.01 mg for the first and second cohort, respectively.

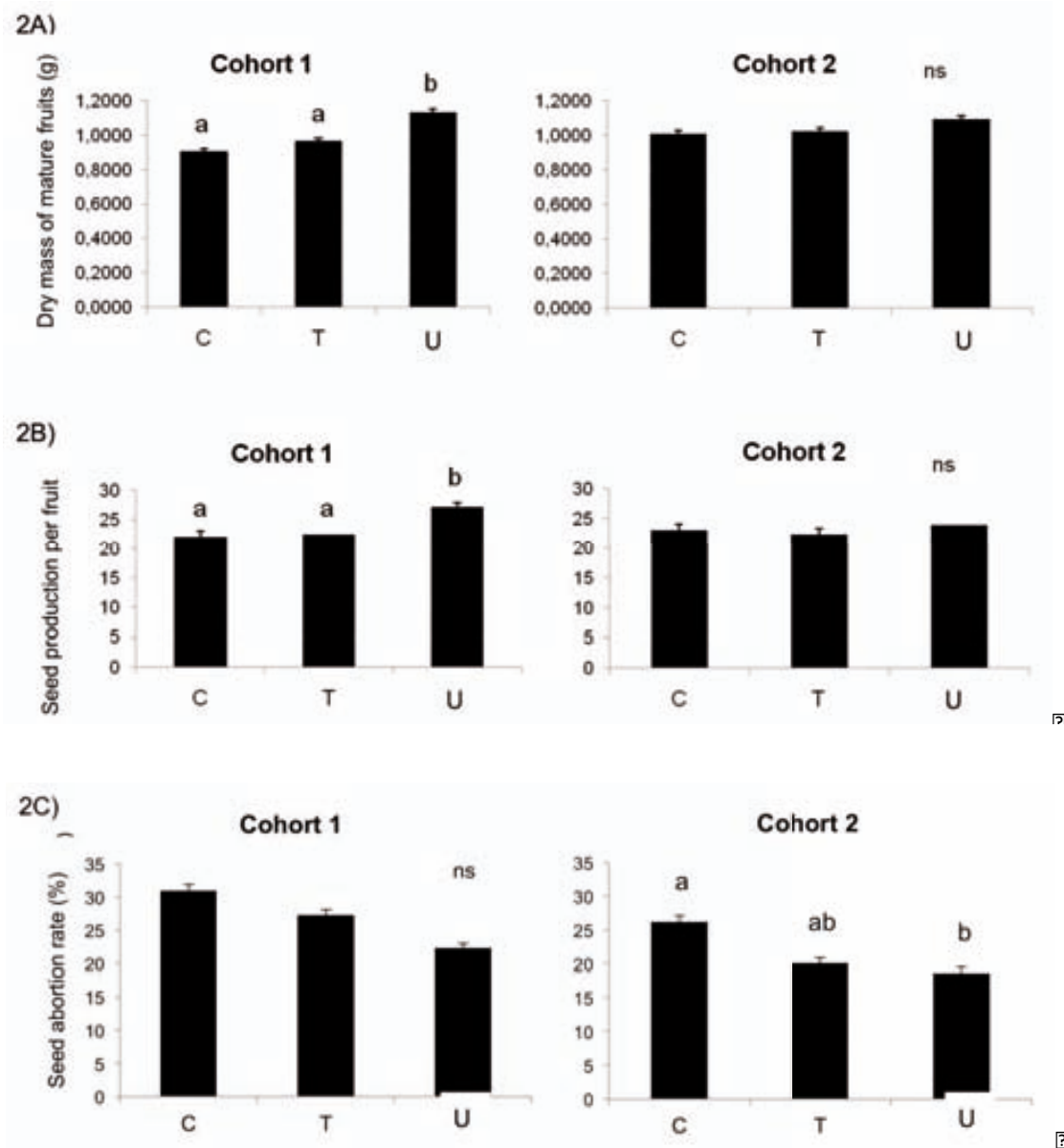
3.4. Stand-level production

There was no significant difference among treatments in the percentage of strawberry trees that flowered for either of the two cohorts ($69\pm 3\%$ and $58\pm 6\%$ for the first and second cohort, respectively), nor was there a difference among treatments in the percentage of strawberry trees that bore fruit for either of the two cohorts ($63\pm 4\%$ and $44\pm 5\%$ for the first and second cohort, respectively) (Table 1). The dry biomass of mature fruits per hectare and the number of developed seeds per hectare were similar among treatments (Table 1) for both cohorts (for the first and second cohort, respectively: 21.8 ± 3.5 kg/ha and 4.2 ± 1.5 kg/ha in mature fruit biomass and $516,818\pm 91,043$ and $92,475\pm 34,032$ developed seeds/ha).

Table 1. Effects of the treatments (C, T, and U) on strawberry tree reproduction for the two study cohorts. The F values, significance values (p), and degrees of freedom were taken from the general linear mixed model ANOVA tables. Significant p-values ($p < 0.05$) are in bold. There were two degrees of freedom for all the analyses.

Source of variation		Treatment effect	
		F	p
<i>Individual-level production</i>			
Number of flowers per individual	Cohort 1	1.3	0.343
	Cohort 2	3.0	0.125
Number of mature fruits per individual	Cohort 1	0.0	0.965
	Cohort 2	0.2	0.853
Flower success rate (%)	Cohort 1	0.2	0.844
	Cohort 2	0.2	0.821
Fruit maturation rate (%)	Cohort 1	0.7	0.522
	Cohort 2	1.0	0.406
<i>Fruit quality</i>			
Dry mass of mature fruits (g)	Cohort 1	6.2	0.034
	Cohort 2	1.3	0.332
Number of developed seeds per fruit	Cohort 1	6.2	0.034
	Cohort 2	2.4	0.175
Per-fruit seed abortion rate	Cohort 1	2.3	0.176
	Cohort 2	5.6	0.043
Dry mass of developed seeds (mg)	Cohort 1	1.2	0.373
	Cohort 2	1.6	0.282
<i>Stand-level production</i>			
Percentage of trees that flowered	Cohort 1	0.5	0.642
	Cohort 2	2.0	0.210
Percentage of trees that bore fruit	Cohort 1	1.4	0.315
	Cohort 2	0.3	0.774
Dry biomass of mature fruits (kg/ha)	Cohort 1	0.3	0.773
	Cohort 2	0.0	0.964
Density of developed seeds (seeds/ha)	Cohort 1	0.1	0.929
	Cohort 2	0.2	0.808

Figure 2. Mean (\pm standard error) of A) dry mass of mature fruits, B) seed production per fruit and C) per-fruit seed abortion rate for the different plot treatments (C, control; T, selective thinning of resprouts; U, selective thinning of resprouts plus mechanical clearing of understory) for the two study cohorts (1: 2007-2008; 2: 2008-2009). Different letters indicate significant differences were present (post-hoc Tukey test; $p < 0.05$).



4. Discussion

This study shows that selectively thinning resprouts in coppices of strawberry trees that have regenerated after a fire does not negatively affect fruit production at the individual- or stand-level. In fact, the selective thinning of resprouts, when accompanied by understory clearing, may even improve fruit quality in the short term at the level of individual trees, probably because this combined treatment releases resources and thus reduces resource competition. Strawberry trees that experienced both selective thinning and understory clearing produced fruits that had much higher dry masses and that contained more developed seeds than trees on control plots or trees that experienced only selective thinning. Therefore, in the short term and at the individual level, the resources released by the combined treatment and the subsequent reduction in competition sufficed to stimulate resprout growth (Quevedo et al., 2013) without compromising reproduction.

Although the treatments did not increase fruit production per individual, they did improve fruit quality (as evidenced by the greater fruit dry mass and per-fruit seed production on treatment plots), which implies that they could also enhance the species' ability to establish new seedlings. Strawberry tree fruits are dispersed via endozoochory (Herrera, 1998), and fruit size is one of the main features used by birds and mammals when choosing food (Debussche and Isenmann, 1989; Herrera et al., 1994). Therefore, fruits of greater mass may be more attractive to wildlife and more likely to be dispersed. This fact, and the fact that fruits produced by trees in the treatment plots contained more, unaborting seeds, may mean that the treatments used in this study could lead to a greater number of seeds germinating and thus higher levels of strawberry tree establishment. The finding that the treatments failed to enhance individual fruit production contrasts with findings from studies on other Mediterranean species that coexist with the strawberry tree. For instance, in the Aleppo Pine (*Pinus halepensis*), it has been shown that selective thinning in high-density areas increases the number of new cones produced per individual (González-Ochoa et al., 2004; Verkaik and Espelta, 2006). In oak species, selectively thinning resprouts can lead to a greater production of acorns per individual (*Quercus rubra*, Healy et al., 1999; *Q.*

illex and *Q. x cerrioides*, Espelta et al., 2007). At the same time, the treatment plots and the control plots had similar levels of dry mature fruit biomass per hectare (i.e., stand-level fruit quality), which contrasts with the treatment-induced increase in fruit quality observed at the individual level. Unfortunately, we do not have a clear explanation for these treatment-related scale-specific differences in fruit quality. Although tree density was similar among treatments and there were no significant differences among treatments in the percentage of trees that bore fruit and mean per-tree fruit production, a slightly higher percentage of trees produced fruits in the control plots (mean±SE: 72.5±3.4, 64.2±9.5, and 70.8±2.9 for the C, T, and U plots, respectively, in 2008). However, there was no significant difference among treatments when production was expressed in terms of fruit biomass per hectare (calculated from the above means). At any rate, taken together, the results of this study strongly suggest that selective thinning and selective thinning combined with understory clearing do not negatively affect strawberry tree reproduction, nor are they likely to negatively affect seedling establishment.

The differences between the treatment plots and the control plots that were detected for the first cohort were largely absent in the second cohort. Moreover, flower production and fruit production were much lower overall for the second cohort (Figure 1). This effect was probably due to the higher levels of caterpillar (*Euproctis chrysorrhoea* L.) infestation experienced by the second cohort, especially in plots that had been both selectively thinned and cleared of understory vegetation. Young caterpillars of this species, when they are in embryonic diapause, feed on the leaves of the strawberry tree (Frago et al., 2010). This fact might explain why plots that experienced selective thinning and understory clearing were more infested with caterpillars: resprout growth was stimulated the most in these plots over the medium term (Quevedo et al., 2013). Higher growth rates mean more tender leaves and shoots, which are the caterpillar's main food source (Conesa, 2000; Kment and Bañar, 2008). Consequently, it seems that this caterpillar adversely affected the strawberry tree's ability to reproduce. Other caterpillars have similar impacts on other tree species; for instance, *Tortrix viridana* L. negatively affects *Quercus illex* (Soria and Notario, 1990) and *Thaumetopoea pityocampa* negatively affects *Pinus*

sylvestris ex. nevadensis (Hódar et al., 2003). Therefore, although the combined treatment may harm strawberry trees by increasing infestation levels, it does not have an apparent effect on growth (Quevedo et al. 2013); it might, however, have been responsible for the second cohort's diminished ability to reproduce. Even though the plots that experienced the combined treatment had the greatest infestation levels, their fruit quantity and fruit quality did not differ from those on control plots. If the caterpillars had not been present, it is possible that the combined treatment would have enhanced reproduction in the second cohort even more.

The fact that the treatments did not reduce strawberry tree flower and fruit biomass at the stand level has two key implications. First, the way in which these treatments enhance strawberry tree reproduction has implications at the population scale. By promoting seedling establishment within the community or in nearby areas through animal-vectored fruit dispersal (Debussche and Isenmann, 1989; Herrera et al., 1994; Aparicio et al., 2008; Rost, 2012a), these treatments help reincorporate genetic variability into the population. Second, the fauna in the area have access to a guaranteed key food resource, especially in autumn, when there are few flowering plants (Picó and Retana, 2001). Strawberry tree flowers are an important source of pollen and nectar for different groups of pollinators that are active during the colder parts of the year; they are especially important for honey bees (*Apis mellifera*L.) and bumble bees (*Bombus terrestris*L.) (Herrera, 2004; Rasmont et al., 2005). Moreover, the fleshy fruits produced by *Arbutus unedo*, which contain high levels of carbohydrates, such as sugars, also act as an important food resource (Ayaz et al., 2000; Barros et al., 2010), especially for mammals (Virgós et al., 2010) and birds (Herrera, 1998).

In this study, we demonstrated that selective thinning and selective thinning accompanied by understory clearing, even when applied 12 years after a fire, do not negatively affect strawberry tree reproduction and may even somewhat enhance it. Consequently, these post-fire management practices, which play a key role in improving the vertical structure of strawberry tree coppices and reducing the risk that fire will spread within them (Quevedo et al., 2013), are suitable ways of managing areas dominated by *A. unedo* L. following fire,

especially from a reproductive perspective. They may also be suitable for other resprouter species with similar traits.

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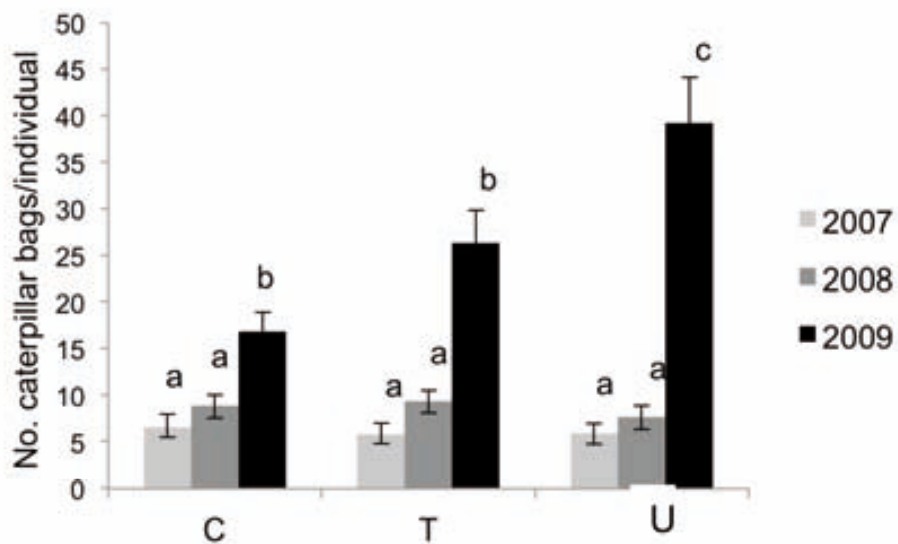
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Appendix 1. Mean (\pm standard error) infestation levels of *Euproctis chrysorrhoea* L. caterpillars on individual *A. unedo* trees for the different plot treatments (C, control; T, selective thinning of resprouts; U, selective thinning of resprouts plus mechanical clearing of understorey). Different letters indicate significant differences were present (post-hoc Tukey test; $p < 0.05$).



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□

CHAPTER 4

Post-fire selective thinning of *Arbutus unedo* L. coppices keeps animal diversity unchanged: the case of ants

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Post-fire selective thinning of *Arbutus unedo* L. coppices keeps animal diversity unchanged: the case of ants

Abstract/Resum

- Context: In Mediterranean area different post-fire management strategies are used for coppices of resprouting species to promote a more regular forest structure, enhance plant growth, and reduce fire risk. However, the effects of these management treatments on forest-associated fauna are unknown, which in turn could be limiting their beneficial effects.
- Aims: To determine whether forest management of a recently burned area dominated by a vigorous resprouting tree species (*Arbutus unedo* L.) affects ant community.
- Methods: Ant communities, sampled using pitfall traps, were examined from unmanaged and selective thinning coppices of *A. unedo*. Ants are here used as bioindicators of ecosystem health and surrogates for other animal groups.
- Results: Very limited effects of these post-fire management strategies on the structure and composition of ant community were found. The lack of effects could be due to: the reported small changes in physical conditions among treatments; and, the most sensitive ant species to these post-fire management treatments might be the same ones affected by fire and, consequently, the ant species that would potentially be affected most were no longer in the study area.
- Conclusion: The lack of any significant effects caused by this post-fire management practices on the associated fauna of *A. unedo* coppices points out the suitability of these treatments in these circumstances.

Keywords: Ants, Biodiversity conservation, Mediterranean, Post-fire management, Selective thinning, Strawberry tree

A la Conca Mediterrània, diferents estratègies de gestió forestal són utilitzades en bosquines d'espècies rebrotadores per promoure una estructura més regular, millorar el creixement de l'espècie, i reduir el risc d'incendi. De totes maneres, els efectes d'aquests tractaments de gestió sobre la fauna forestal associada són desconeguts, fet que podria limitar els seus efectes beneficiosos. L'objectiu d'aquest treball va ser determinar si la gestió forestal d'una zona recentment cremada i dominada per una espècie arbòria amb una rebrotada molt vigorosa (*Arbutus unedo* L.) afecta les comunitats de formigues. Les comunitats de formigues, mostrejades utilitzant trampes de caiguda, van ser examinades tant de bosquines no gestionades com de bosquines on es va aplicar una selecció de rebrots d'*A. unedo* (amb o sense desbrossada del sotabosc). Les formigues van ser emprades com a bioindicadores de la salut de l'ecosistema i com representants d'altres grups faunístics. Els resultats van mostrar uns efectes molt limitats d'aquests tractaments forestals postincendi en l'estructura i en la composició de les comunitats de formigues. La manca d'efectes podria ser degut als petits canvis detectats en les condicions físiques entre tractaments; o per altra banda, les espècies de formigues més sensibles a aquests tractaments postincendi podrien ser les mateixes que les afectades pel foc i conseqüentment, les espècies de formigues que potencialment podrien ser afectades ja no estaven presents a la nostra àrea d'estudi. La manca d'efectes causats per les pràctiques de gestió forestal postincendi sobre la fauna associada a les bosquines d'*A. unedo* assenyala la sostenibilitat d'aquests tractaments en aquestes condicions.

1. Introduction

Logging and wildfires are two forest disturbances that promote major shifts in vegetation cover, which in turn lead to variations in microhabitat conditions and resource availability for associated fauna. Consequently, these changes may affect the composition and structure of animal communities. Accordingly, decreases in the diversity of forest fauna have been documented following different types of forest management activities (e.g. Thiollay 1997; Kavanagh and Webb 1998; Davis et al. 2001). In the case of wildfires, direct effects on forest fauna have also been reported, causing both mortality and fleeing, and indirect effects, such as the modification of forest habitats (e.g. Arnan et al. 2007; Rost et al. 2012). Thus, species that survive a fire or return after a fire may have difficulties living in the new conditions. On the other hand, there are species that are capable of reestablishing in these modified habitats that have different characteristics compared to before the fire (e.g. McCulloch et al. 1998; Rodrigo et al. 2008, Arnan et al. 2013b).

Wildfires, which are increasingly more frequent and intense (Piñol et al. 1998), are a major source of disturbance that has profound impacts on forests in the Mediterranean basin (Rodrigo et al. 2004). In many burned areas, different forestry management treatments are often applied after a fire, either to harvest burned wood (Rost et al. 2012; Quevedo et al. 2013) or to encourage faster regeneration. For example, selective removal of resprouts may be applied in areas dominated by species with a great capacity for post-fire resprouting (Quevedo et al. 2013). In this context, post-fire forest management practices can be understood as a second disturbance to the ecosystem, which could, in turn, affect the fauna that has established in the burned area and thus hinder their recovery (Apigian et al. 2006).

Ants are among the most diverse, abundant, and ecologically significant organisms on Earth (Hölldobler and Wilson 1990). They can modify the abiotic and biotic properties of their environment by performing a variety of ecological functions (e.g. Hölldobler and Wilson 1990; Zelikova et al. 2011). Consequently, ants are considered to be crucial components of most terrestrial ecosystems

(Hölldobler and Wilson 1990; Lach et al. 2010). Ant communities are often used as a bioindicators of ecological change, arising from natural or anthropological disturbances (e.g. Vasconcelos et al. 2000; Andersen and Majer 2004; Hoffmann and Andersen 2003; Nakamura et al. 2007). In the case of fire, ant community recovery post-fire depends on both the direct mortality caused by the high temperature as well as by the changes in the vegetation cover caused by fire (Andersen et al. 2006; Arnan et al. 2006, 2007; Rodrigo and Retana 2006; Parr and Andersen 2008). Therefore, fires can generate changes in the composition of ant communities (Andersen et al. 2009; Arnan et al. 2006), including species richness and diversity, which in some cases increase (Mackay et al. 1991; Andersen et al. 2009), in others decrease (York 2000; Rodrigo and Retana 2006), or, as in some Mediterranean ecosystems, are not affected by fire (Arnan et al. 2006).

It is also known that forest management practices can affect ant communities (Vasconcelos et al. 2000; Dunn 2004; Gómez and Abril 2011), since in the short term it often implies a decrease in vegetation cover and may also alter food availability. However, as far as we know, there is no information available regarding whether post-fire salvage logging, which involve a modification of forest condition, affect the recovery of ant communities. This is of particular interest for species that inhabit closed canopy forests, as these species are the most affected by fire (York 2000; Parr and Andersen 2008; Andersen et al. 2006, 2009; Arnan et al. 2006; Rodrigo and Retana 2006). Therefore, it is important for the conservation of areas affected by fires, as is the case of the Mediterranean basin, to determine whether the post-fire forest management treatments often applied to large areas (Quevedo et al. 2003) can be considered a threat to the recovery of ant communities after a fire event.

The aim of this work is to determine whether forest management of a recently burned area dominated by a vigorous resprouting tree species affects the local fauna. In particular, this study focuses on the effects of the application of selective thinning of resprouts and the selective thinning of resprouts plus mechanical clearing of the surrounding vegetation on ant communities established after fire in a strawberry tree (*Arbutus unedo* L.) coppice. We test the

following hypotheses: a) Post-fire forest management treatments on the strawberry tree coppices will change the structure and composition of ant communities, since environmental conditions will also change; and b) The more intense the forest management treatment, the greater the changes to ant communities. This is the first study that analyzes the effects of selective thinning of resprouts species on fauna in a post-fire scenario in a Mediterranean area, where wildfires have a great impact on biota.

2. Materials and methods

2.1. Study area

This study was undertaken in the north of the Baix Llobregat region (41 ° 35 ' N, 1 ° 52 ' E, Catalonia, NE Iberian Peninsula), in a 240 hectares area affected by three canopy wildfires in recent years (1985, 1986 and 1994). Before these fires occurred, this area was covered by Aleppo pine (*Pinus halepensis* Mill.) forests and, after the fire events, developed forested areas of Strawberry trees (Arnan et al. 2013a). The study area is located between 390 and 500 m above sea level and in a region with a dry Mediterranean climate (according to Thornthwaite's humidity index), with a mean annual temperature of 13.5 ° C and a mean annual precipitation of 650-700 mm.

The strawberry tree (*Arbutus unedo* L.) is a sclerophyllous tree species that belongs to the family of Ericaceae. In our study area, the strawberry tree forests had a density of 1265 ± 625 individuals/ha and a number of 15494 ± 7154 resprouts/ha, before treatments (Quevedo et al. 2013). In general, the strawberry tree has very vigorous resprouts after a disturbance (such as a cut or a fire) and can develop up to 60 resprouts/stump (Quevedo et al. 2013). This vigorous resprouting response involves a slow growth of stems due to the high competition for resources at the individual level. In these conditions, the lack of forest management promotes a high horizontal continuity and, as consequence, a high fire risk (Quevedo et al. 2013).

2.2. *Experimental design and application of post-fire experimental treatments*

In 2006, we established four sampling zones in the study area (6.5 ha each one, hereafter, blocks), distant 250-400 m between them, with a high density of *A. unedo* vigorously resprouted after the last fire (1994). In each block, we randomly chose three plots of 50 m x 50 m of similar slope (10 to 30%) and aspect (southwest) (distance between plots: 40-125 m). All the selected plots showed similar forest structural characteristics. For a more detailed description of spatial position and characteristics of plots, see Quevedo et al. 2013).

In the winter of 2006-2007, the three plots within each block were assigned one of three management strategies at random: 1) control strategy (henceforth C), in which nothing was done to the plot; 2) a selective thinning treatment (henceforth T), in which resprouts were selectively removed with electric scissors (Electrocoup F3005) from all of the strawberry trees in the plot – 1 out of 5 resprouts was retained per individual (preserving the largest diameter resprouts); 3) a selective thinning plus understory clearing treatment (henceforth U), in which the same selective thinning procedure was applied in tandem with the mechanical clearing of all the understory vegetation of the plot (for more details see Quevedo et al. 2013).

2.3. *Field sampling*

2.3.1. *Characterization of the strawberry tree forest structure*

Vegetation structure is an important predictor of the structure and composition of ant communities (Retana and Cerdá 2000; Lassau and Hochuli 2004; Arnan et al. 2007). In order to characterize the type of habitat, we established a 40 m x 40 m physical grid in each plot, with each cell measuring 1x1m. Each of the 1600 cells was visually assigned one of the following three types of habitats: 1) *Arbutus* cover, when the majority of the cell was covered by strawberry tree; otherwise, 2) shrub cover, if plant coverage had a height between 1.20 m to 3 m; and 3) herbaceous cover, if that coverage was lower than 1.20 m or if no

vegetation cover was present (bare and rocky soil). Then, we established the habitat composition of each plot as the percentage of each type of habitat.

To characterize the abiotic environment of each habitat type, we measured different environmental variables during each season of 2008:

a) Soil moisture. Three soil samples were taken from each plot, one for each of the three habitat types (total of 36 samples). Each sample, in turn, was formed by three subsamples of the same habitat, taken within the same plot. Soil samples were taken with a manual metal probe (1500 cm³), placed in a sealed plastic bag, weighed in the laboratory, set to dry at 105 °C for 24 hours, and subsequently reweighed. The moisture content of the soil was obtained as the difference between the wet and dry weight.

b) Percentage of shade provided by vegetation. This was determined by measuring the photosynthetically active radiation (PAR) with a Decagon Sunfleck SF 40 Ceptometer (Delta Devices, Cambridge, UK) at the time of maximum sunlight, above and below the vegetation canopy. Measures were obtained in each plot and habitat type directly (5 repeats per habitat) (total of 36 samples). Then, the percentage of shade in each point was calculated as the percentage of difference between the two PAR values (above and below) in respect to the above PAR value.

c) Temperature. Temperature at ground level was obtained with continuous temperature registers (Stow Away Tidbit Temp Logger; Onset Computer Corp., Pocasset, MA) placed at ground level in each plot and habitat type. The sensors recorded temperature data every 30 minutes for a week. Since the sensors measured a maximum limit of 38 °C, the percentages of temperature data values below 20 °C and above 35 °C were used, representing the values out of the range of temperatures in which Mediterranean ants are active (Retana and Cerdá 2000).

2.3.2. *Ant sampling*

We used pitfall traps to measure the ground ant composition and abundance of each plot. Sampling was conducted in 2008, 2009 and 2010, and in May and July, when ant activity is highest in Mediterranean ecosystems (Cros et al. 1997).

Pitfall traps were 7.5 cm in diameter, 9.5 cm deep, plastic vials partially filled with water, soap and salt. A 4 x 4 grid of traps with 5 m spacing was established in each plot. The traps were placed in the center of the plot to minimize edge effects and were operated for one week for each sampling period. The contents of the 16 traps of each plot were lumped together to obtain a single sample per plot. The samples from the two sampling periods (May and July) from each plot and year were pooled for analysis, so that we had one sample for the whole activity period of ants (i.e. total number of ants per 16 traps across 14 days by each year). The ants were then sorted in the laboratory and identified to the species level according to Bernard (1968) and Espadaler (1990), supplemented with the identification of some specimens by Dr. Xavier Espadaler.

2.4. Data analysis

The effects of forest management (C, T and U) on habitats composition were analyzed using blocked univariate ANOVAs, where treatment was the main factor and block the random factor. We conducted a separate analysis for each habitat type (*Arbutus* cover, shrub cover and herbaceous cover). The effects of forest management and habitat type on the different abiotic variables (soil moisture, PAR, and temperature values below 20 °C and above 35 °C) were also analyzed using blocked univariate ANOVAs, where treatment, habitat and season were the main factors, and block the random factor. We conducted a separate analysis for each abiotic variable and season (winter, spring, summer and fall). We conducted post hoc comparisons using the Tukey test when the differences for the main factors were significant. The percentages of each habitat type and the PAR were arcsine-square root transformed in order to achieve data homoscedasticity. The statistical package SPSS (SPSS Inc. 2006) was used for these analyses.

To investigate the composition and structure of communities of ants in each plot, we computed the following indexes: (a) overall abundance of ants in pitfall traps; (b) species richness (S); (c) Shannon diversity index ($H' = -\sum_{p=1}^S p_i \ln p_i$, where p_i

is the proportion of workers of the i^{th} species in traps and S is the number of species); and (d) the index of numerical dominance of the most abundant ant species ($ID = 100 (y_1/y)$, where y_1 is the abundance of the most abundant species and y is the overall abundance of all ant species in the plot). Each ant species was assigned to a particular trophic guild according to its diet, as determined from Arnan et al., 2013b. We established the following guilds: liquid food (nectar and/or honeydew), liquid food and insects, only insects, insects and seeds, and only seeds. We analyzed the changes in the structure of ant community and the proportion of individuals belonging to the different dietary guilds among treatments and years by using mixed linear models, where treatment, year and their interaction were the main factors, and block and plot the random factors. Overall abundance was log transformed, while the numerical dominance index and the relative abundance of each trophic guild were arcsine-square root transformed in order to achieve homoscedasticity. The statistical package R (version 2.13.0) was used in these analyses (function “lme” in package “nlme”).

Significant differences in the ant community composition between treatments were determined with multivariate analysis using the program Primer 5.1.2 (Clarke and Gorley 2006). Ant abundance per plot was log transformed to obtain homoscedasticity. First, plots were ordered using non-metric multidimensional scaling (NMDS), based on the Bray-Curtis similarity matrix. Then, analysis of similarities (ANOSIM) was used to test for significant differences in ant community composition related to forest management treatment (C, T and U). Ant species that occurred in less than 3 plots and that had a relative abundance of <1% were excluded from the analysis. Finally, similarity percentages (SIMPER) were used to identify the ant species that contributed most to differences in community composition. All these analyses were conducted for each year (2008, 2009 and 2010).

3. Results

3.1. Habitat composition

The U plots showed significant differences in habitat composition with respect to the C and T plots. These had a lower percentage of shrub cover ($F=30.5$; $p<0.0001$: $6.2\pm 1\%$, $19.2\pm 2.0\%$ and $27.5\pm 9.6\%$ for the U, C and T plots, respectively) and a greater percentage of herbaceous cover ($F=12.8$; $p=0.007$: $64.5\pm 8.9\%$, $39.6\pm 10.0\%$ and $8.4\pm 38.4\%$ for the U, C and T plots, respectively), with respect to the other two types of plots. There were no significant differences in the habitat composition between plots C and T ($p>0.05$). There were also no significant differences for the *Arbutus* cover category between plots of different treatments ($p>0.05$, $41.2\pm 11.1\%$, $34\pm 5.8\%$ and $29.3\pm 8.1\%$ for the C, T and U plots, respectively). According to the different composition of habitats in the plots and to differences in the physical characteristics of habitats (Table 1), plots showed only small differences between treatments in soil moisture in spring and fall and in shadow in summer, but no differences for temperature (Figure1).

Table 1. F values of the blocked univariate ANOVA tests for each season and for each variable that we used to characterize habitats (soil moisture (%), percentage of shade exerted by vegetation (PAR), percentage values of temperature ground level below 20 °C (T20) and percentage above 35 °C (T35)) in the treatment (T), habitat (H) and block (B). Significant values ($p \leq 0.05$) are indicated in bold. * Analysis not available due to lack of data.

Source	Winter			Spring			Summer			Fall		
	T	H	B	T	H	B	T	H	B	T	H	B
Soil moisture (%)	2.4	0.8	4.1	3.9	1.2	2.1	0.1	0.6	1.1	5.6	1.3	3.1
Shade (%)	1.9	55.5	5.7	0.1	86.5	5.4	4.3	32.9	12.3	0.1	24.6	9.9
T20	0.6	3.4	3.4	0.7	4.7	43.2	1.2	2.0	5.9	0.7	2.3	58.0
T35	*	*	*	2.0	1.0	1.3	1.7	1.9	3.9	0.7	2.4	2.7

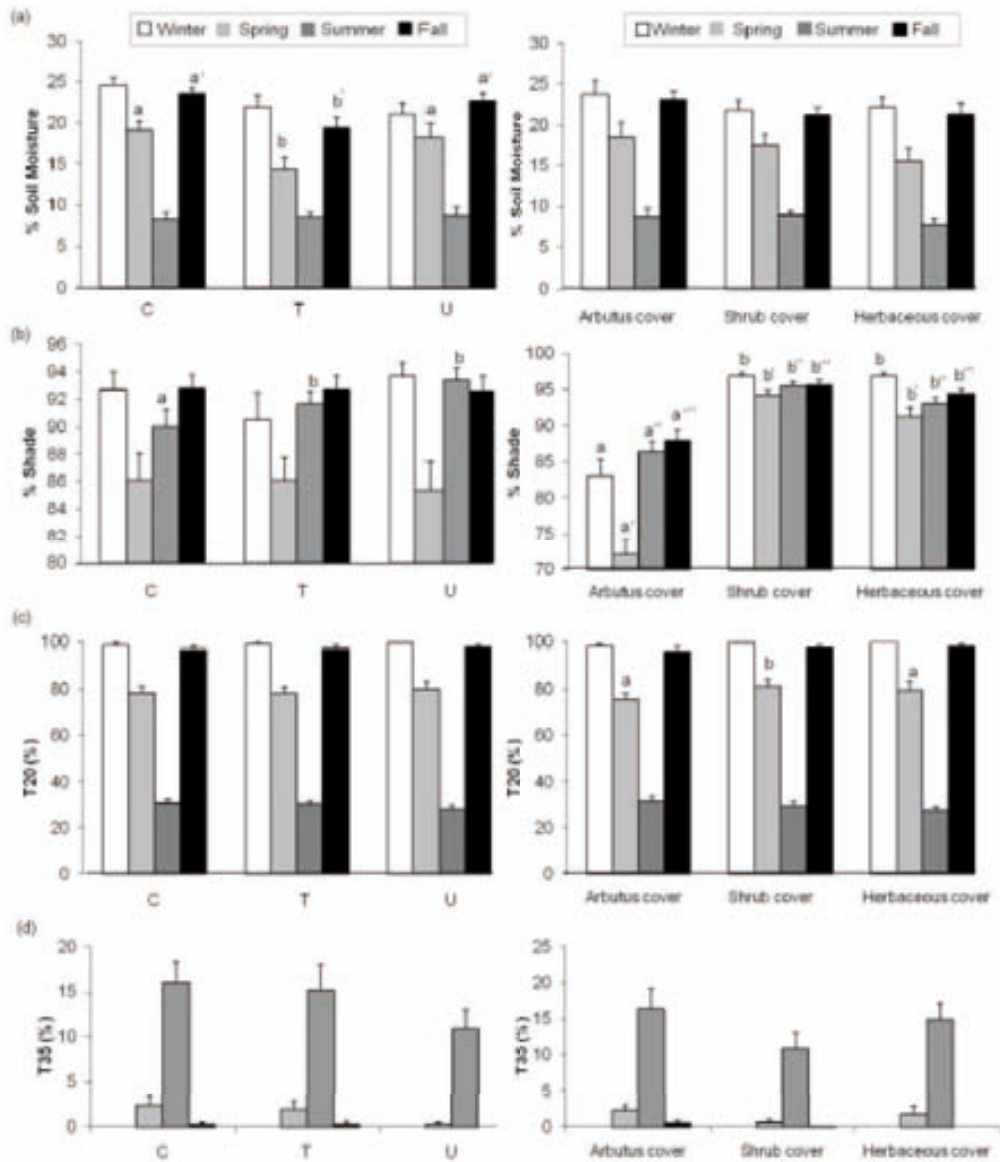


Figure 1. Environmental variables measured for each forest management treatment and habitat for each season: winter (white bars), spring (light gray bars), summer (dark gray bars) and fall (black bars): a. percentage of soil moisture, b. percentage of shade, c. percentage values of temperature below 20° C (T20), d. percentage values of temperature above 35° C (T35). The vertical bars indicate standard error (SE) of the mean. Different letters indicate significant differences between categories, according to the Tukey post-hoc tests. Abbreviations: C = control, T = selective thinning, U = selective thinning plus understory clearing

3.2. Ant community structure

In total, 38159 worker ants from 31 species were collected (12741, 9600 and 15818 from years 2008, 2009 and 2010, respectively) (Table 2). Species richness varied between 19 and 22 in the C plots, between 22 and 25 in the T plots, and between 20 and 22 in the U plots.

The mixed linear models showed no effects of treatments, years or their interaction on species richness, the Shannon diversity index (H') and the index of numerical dominance of the most abundant ant species (Table 3). Significant differences were found between the years 2008 and 2009, as well as between 2009 and 2010, in terms of total ant abundance (467 ± 61 , 356 ± 88 and 705 ± 158 ants per plot for the years 2008, 2009 and 2010, respectively) (Table 3).

3.3. Ant community composition

The ANOSIM analyses showed only a slight effect of treatments on ant community composition in the long term (year 2010; Global $R = -0.238$, $p = 0.014$, Figure 2; $p > 0.05$ for the years 2008 and 2009). The only significant differences were between the C and U plots (post-hoc analyses: $p = 0.029$ between the C and U plots; $p = 0.114$ between the C and T plots; and $p = 0.057$ between the T and U plots). Nevertheless, Figure 2 shows no clear separation of treatments. In fact, the negative value of the global R statistic indicates a very large variability within plots of the same treatment, but very low variability between treatments. So, despite the fact that it seems that there were significant differences between ant community from C and U plots, they were very slight and difficult to assess given the high variability within treatments. The SIMPER analysis revealed that the species that contributed to the slight dissimilarity between C and U plots were *Messor capitatus* (12.07%), *Formica subrufa* (10.52%), *Formica gagates* (10.17%) and *Pheidole pallidula* (8.78%).

Table 2. List of ant species and number of worker ants collected in the plots of the three treatment intensities during the three years of study (2008, 2009 and 2010). The intensities of treatment are: control, C, selective thinning, T, and selective thinning plus understory clearing, U. Parentheses indicates the number of plots in which each species appears for each year and intensity of treatment. Abbreviations are the type

	1 (1)	19 (3)	11 (2)	20 (2)	1 (1)	9 (2)	16 (1)
(4)	726 (4)	504 (4)	227 (4)	723 (4)	193 (4)	163 (4)	502 (4)
	2 (2)	1 (1)	-	3 (2)	2 (2)	1 (1)	1 (1)
	36 (4)	19 (3)	12 (3)	34 (3)	17 (4)	13 (4)	19 (4)
	29 (4)	9 (3)	6 (4)	19 (4)	27 (3)	25 (4)	30 (4)
	1 (1)	-	1 (1)	7 (2)	-	1 (1)	-
	2 (1)	-	-	1 (1)	-	-	-
	88 (2)	237 (4)	226 (4)	190 (3)	165 (3)	479 (3)	250 (4)
	235 (2)	572 (3)	323 (2)	189 (2)	1228 (1)	1229 (2)	755 (2)
	6 (2)	80 (4)	20 (2)	47 (3)	2 (2)	1 (1)	1 (1)
	441 (2)	432 (3)	480 (3)	507 (3)	1088 (2)	2050 (2)	3547 (2)
	17 (3)	1 (1)	14 (1)	4 (2)	1076 (2)	69 (3)	476 (2)
	-	-	1 (1)	2 (2)	-	-	3 (1)
(4)	380 (4)	511 (4)	201 (4)	346 (4)	343 (4)	289 (4)	556 (4)
	-	1 (1)	-	-	-	-	1 (1)
	101 (4)	155 (4)	73 (4)	116 (3)	162 (3)	82 (3)	132 (3)
	3852	3876	2177	4683	4605	4994	7283
	22	23	22	25	20	22	21

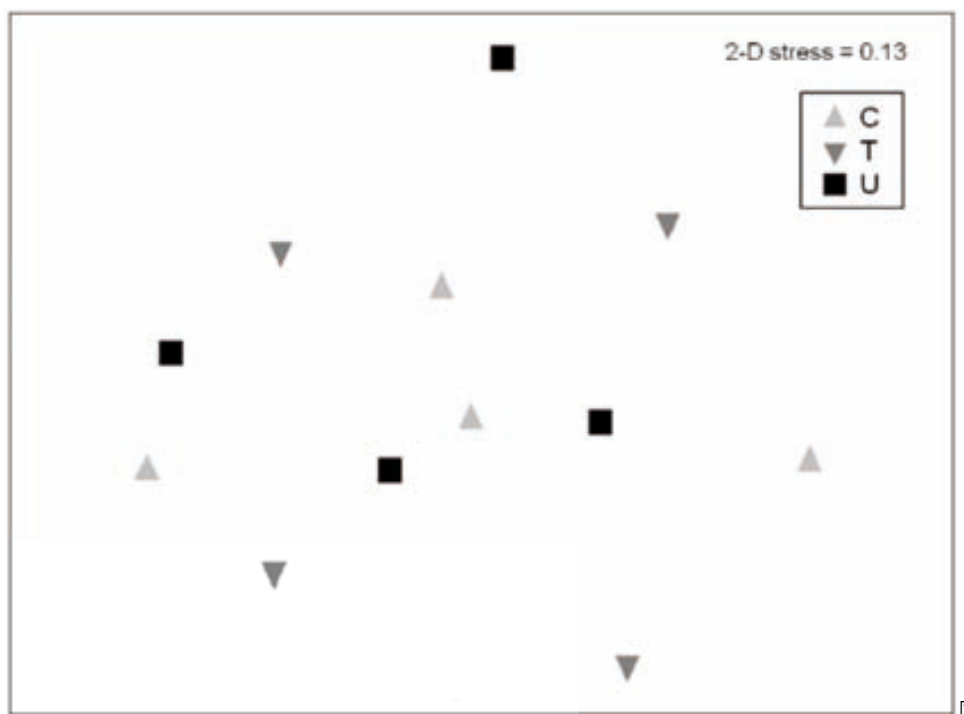


Figure 2. Non-metric multidimensional scaling (Nmds) ordination in two dimensions of plots based on ant abundance data in 2010. Abbreviations: C, control; T, selective thinning, and U, selective thinning with understory clearing

3.4. Ant dietary guilds

The relative abundance of each ant dietary guild was not significantly different between treatments or years. Their interaction was not significant either (mixed linear models, $p > 0.05$ in all cases).

Table 3. Effect of treatment (forest management: C, T and U) and year (2008, 2009 and 2010) on variables describing structure of ant communities. F values, significance (p), and degrees of freedom (g.l.) comes from the ANOVA tables of the mixed linear models. Significant values ($p < 0.05$) are indicated in bold.

Source of variation	df	Total ant abundance		Richness		Shannon diversity index (H')		Numerical dominance	
		F	p	F	p	F	p	F	p
Treatment	2	1.2	0.374	1.7	0.256	1.8	0.247	2.3	0.185
Year	2	9.3	0.002	1.6	0.236	0.3	0.777	0.2	0.828
Treatment*Year	4	0.4	0.819	1.4	0.278	0.1	0.981	0.1	0.961

4. Discussion

This work has demonstrated that there is virtually no effect of post-fire selective thinning on the structure of ant communities inhabiting the *A. unedo* coppices of our study area. The applied forest management treatments did not cause any impact, either positive or negative, on the total abundance, species richness, diversity, dominance, or the composition of ant community, neither in the short or the medium term (Table 3, Figure 2). This result contradicts numerous works, much from the tropics of Amazon, where it has been found that the application of selective thinning leads to a negative impact on the species richness or the composition of ant species (e.g.: Roth et al. 1994; Vasconcelos et al. 2000; Dunn 2004; Nakamura et al. 2007; Palladini et al. 2007) and other research on faunal groups also used as bio-indicators, as for example, beetles (Davis et al. 2001), birds (Thiollay 1997), small mammals, reptiles and amphibians (Kavanagh and Webb 1998). However, there are other studies that show similar results to those provided in this work, that indicate that some forest harvesting does not negatively affect the overall animal biodiversity (Azevedo-Ramos et al. 2006) and have a minimal impact on ant communities (Vasconcelos et al. 2000; Andersen et al. 2009). Several of these works agree that one of the key factors that determine if forest harvesting will have an impact on the richness and/or composition of biodiversity is the intensity, and also the frequency, of treatments (Dunn 2004; Andersen et al. 2009; Gómez and Abril 2011). The more intense the treatment, more change is generated on the vegetation cover, causing a larger impact on the whole ecosystem (Maeto and Sato 2004; Palladini et al. 2007). Nonetheless, we were unable to detect this larger effect in even our most intense treatment, which involved both the selective thinning and the clearing of understory vegetation.

Soil moisture determines ant foraging activity, the abundance of food resources, the suitability of places to make their nests, as well as predation by other species of ants (Wang et al. 2001). Also, ground temperature has a direct effect on nest temperature levels and an indirect effect on food supply via vegetation growth (York 2000). In fact, when we applied selective thinning along with the clearing of the understory, we were able to modify proportions of habitats, resulting in a

greater availability of more open habitats. However there are only small differences in ground temperature between habitats but similar soil moisture, probably due to an early resprouting of the understory vegetation. As a consequence, although this change in open habitat proportion differences between treatments as a whole in physical conditions were very small (Fig 1). So, these small changes were not great enough to affect the structure or the composition of the ant community.

Our results also show that these treatments could not modify the relative abundance of ant dietary guilds in the study area. Although it might make more sense that clearing undergrowth would cause a decrease in food resources, such as seeds generated by the shrub layer and insects that inhabit this environment (termites, springtails, other ants, etc.) (Hölldobler and Wilson 1990; Arnan et al. 2006), this did not yet have any effect or was not important enough to negatively impact ants that feed on these food resources. On the other hand, the shrubs present in our study area have a high resprouting capacity after cutting (*Pistacia lentiscus*, *Quercus coccifera*, *Thymus vulgaris*, *Viburnum tinus*, etc.), so that they recover quickly and with ease.

One possible explanation for the lack of any effect of post-fire forest management treatments on ant community might be that the ant species most sensitive to these forest practices are the same ones that are also most affected by changes in vegetation promoted by fire, i.e., tree-dwelling ant species (Arnan et al. 2013b). Consequently, the ant species that would be most affected by post-fire management treatments were no longer in the study area. As such, post-fire forest management might affect the process of post-fire colonization in the long term, although we did not detect any evidence in the short and midterm.

At any rate, the act of reducing forest cover by selective thinning has minimal effect on the ant community of this area, nor does it if the understory is also removed. This implies that the application of such treatments that have clear advantages in the recovery of the structure of these burned forests and in the decrease of fire risk (Quevedo et al. 2013) may be advisable on a regional basis without any negative effect in the ant fauna. In fact, as these treatments favor the

height and diameter growth of the stands of the strawberry tree (Quevedo et al. 2013), accelerating the recovery of these forests towards more structured and closed, might even encourage, in the medium and long term, the appearance of ant species normally only found in more closed areas (Arnan et al. 2006; Rodrigo and Retana 2006). However, despite the fact that ants can be used as bioindicators of the response of fauna in general (Andersen and Majer 2004; Hoffmann and Andersen 2003), it would be appropriate to check the effect of these treatments on other faunal groups. In this context, analyzing the effect of similar treatments on the faunal community in the medium or long term is important in order to be able to understand the overall effect of these post-fire forest management treatments in the biodiversity of burned areas.

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DISCUSSIÓ GENERAL



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Aquest treball constata, per una banda, un clar augment de la superfície ocupada per les bosquines d'arboç (bàsicament a partir de pinedes de pi blanc amb una recurrència alta de focs), com a conseqüència de canvis en el règim d'incendis forestals existent a Catalunya a les darreres dècades (capítol 1). Per altra banda, davant el desconeixement de com gestionar aquestes bosquines d'arboç rebrotades després del foc, s'ha demostrat que l'aplicació d'una selecció de rebrots, acompanyada o no de la desbrossada del sotabosc, és una gestió adequada per aquestes bosquines, ja que resulta beneficiosa tant en termes de creixement individual, d'estructuració de la massa i la disminució del risc d'incendi (capítol 2), així com en termes reproductius de l'arboç (capítol 3). A més, utilitzant les comunitats de formigues com bioindicadores de la diversitat faunística, hem pogut mostrar que els tractaments no presenten un greuge per la fauna invertebrada (Andersen i Majer 2004, Leal et al. 2010) (capítol 4). Per tant, l'efecte d'aquests tractaments silvícoles és globalment positiu, tant a nivell d'individu com de bosquina, no mostrant efectes negatius en cap cas. Finalment, aquesta tesi també fa una aportació interessant per conèixer amb detall la fenologia reproductiva de l'arboç (capítol 3).

1. Les bosquines d'arboç: una nova coberta forestal en expansió

Habitualment, l'arboç s'ha descrit com a una espècie arbustiva acompanyant en diverses formacions forestals mediterrànies, tals com alzinars, rouredes, suredes, pinedes, estepars i bruguerars (Moro 1995). En el capítol 1 constatem que a Catalunya, el 70% de la superfície de les bosquines d'arboç que han aparegut en els darrers anys provenen de zones que estaven cobertes per boscos de pi blanc (*Pinus halepensis*) i que s'han cremat dos o més vegades. Això ha estat així sempre que el temps entre focs ha estat inferior al necessari per a què el pi blanc hagi pogut desenvolupar llavors viables (Tessler et al. 2014). Per tant, els focs recurrents han provocat que el pi blanc hagi estat incapaç d'establir-se després d'aquests focs, donant pas a la dominància de l'arboç, que es trobava com a espècie acompanyant en el sotabosc. De fet, s'ha pogut observar una reducció de la presència de pinedes de pi blanc i un increment de la presència i la distribució de les bosquines d'arboç a Catalunya. Aquest fet també s'ha constatat amb altres espècies de rebrotadores (per exemple, l'alzina o el roure) (Retana et al. 2002, Espelta et al. 2003, Gracia et al. 2004), però en el cas de l'arboç té molta més rellevància, ja que les cobertes forestals

dominades per ell eren pràcticament inexistentes a Catalunya (MCA 1980), pel que podríem parlar de l'aparició d'una nova coberta forestal (8.662 hectàrees, Departament de Medi Ambient i Habitatge i Universitat de Barcelona 2005) en els paisatges mediterranis del nord-est de la Península Ibèrica.

Donat que es pot pensar que l'actual règim de focs a la Conca Mediterrània es mantindrà i, fins i tot, pot augmentar la recurrència dels mateixos (Moreira et al. 2011, Loepfe et al. 2014), es pot preveure que canvis en l'estructura i la composició del paisatge a una escala regional, consistents en la substitució de masses dominades per pins per bosquines dominades per arboç, es continuaran produint. A més a més, aquesta substitució també es podria donar per espècies amb característiques similars a les de l'arboç, amb una important capacitat de rebrotada i tradicionalment descrites com acompanyants. Per exemple, espècies com *Phillyrea angustifolia*, *Quercus coccifera* o *Pistacia lentiscus* (Sabaté et al. 1990, Manes et al. 2002, Vitale et al. 2007), en determinats escenaris podrien arribar a formar bosquines com a espècies dominants. Per totes aquestes noves cobertes caldria definir quins tractaments silvícoles postincendi cal aplicar, tal com s'ha estudiat per a les bosquines d'arboç, amb la finalitat de millorar-ne la seva estructura, productivitat, reproducció, i reduir el seu risc d'incendi. Aquí presentem, doncs, un marc experimental, provat amb èxit per l'arboç, i que podria ser utilitzat per altres espècies.

2. La selecció de rebrots com a tractament per millorar les bosquines d'arboç

Hi ha una manca de treballs que avaluïn la dinàmica del creixement i la reproducció de l'arboç, així com d'anàlisi de la seva resposta a l'aplicació de tractaments silvícoles, ja que fins ara no s'havia constatat que constituís formacions forestals on fos l'espècie arbòria dominant amb una estructura de bosc adult, del qual se'n poguessin fer aprofitaments de fusta i llenyes. En aquesta tesi hem demostrat com tractaments basats en la selecció de rebrots promouen respostes beneficioses a les bosquines d'arboç, tant a nivell de la seva estructura com del creixement dels individus d'arboç (capítol 2), en la seva reproducció (capítol 3), i no tenen un efecte perjudicial en la comunitat faunística d'aquestes bosquines (capítol 4).

2.1. Estimulació del creixement de l'arboç

L'aplicació d'una selecció de rebrots provoca un augment del creixement en longitud i en diàmetre dels rebrots reservats d'arboç, de manera que són un 48% més llargs i un

60% de major diàmetre que els rebrots dels individus control (capítol 2). Aquests resultats s'obtenen quan la selecció de rebrots s'aplica a nivell individual, independentment de si les parcel·les estan totalment tractades o no. Per tant, els resultats obtinguts són aplicables tant quan ens trobem bosquines dominades per arboç, com quan aquesta espècie es troba com a acompanyant en alzinars, suredes, pinedes, etc., de manera que l'àmbit aplicable d'aquest estudi es pot extrapolar a tota l'àrea de distribució de l'arboç a la Conca Mediterrània. El fet de creuar els tractaments a nivell d'individu i de parcel·la per delimitar a quin nivell es dona l'efecte dels tractaments és una aportació innovadora que fa aquesta tesi, que cap estudi que coneguem en el camp de la gestió forestal fins ara havia incorporat. Aquesta aproximació permet determinar amb exactitud com prioritzar els esforços en la gestió forestal, per tal de minimitzar les despeses de gestió i alhora, optimitzar els beneficis que generen sobre la vegetació tractada.

La resposta descrita és similar a l'obtinguda per *Quercus ilex* i *Q. cerrroides* (Espelta et al. 2003, López et al. 2009), espècies mediterrànies com l'arboç; o per altres espècies rebrotadores típiques de climes més humits, com *Q. coccinea* Muench. (en terres altes de Nord Amèrica), *Q. velutina* Lam. (en ecosistemes dunars) (Cutter et al. 1991) i *Acacia harpophylla* (en boscos oberts plujosos d'Austràlia) (Dwyer et al. 2010). En aquests darrers treballs s'obtenen diferències en el creixement entre intensitats de selecció de rebrots, tot i que aquests tractaments no són comparables als aplicats en aquesta tesi (la selecció de rebrots, acompanyada o no de la desbrossada del sotabosc). El fet que per l'arboç no hi hagi diferències entre tractaments a curt termini i a nivell individual pot ser degut a que la desbrossada del sotabosc no allibera tants recursos com una reducció més intensa de rebrots.

D'altra banda, a diferència dels treballs realitzats amb *Quercus ilex* i *Q. cerrroides* (Espelta et al. 2003, López et al. 2009), on una major intensitat de selecció de rebrots provoca que els nous rebrots induïts siguin més llargs, la rebrotada induïda de l'arboç és independent del fet d'afegir la desbrossada del sotabosc al tractament de selecció de rebrots, tant en número de rebrots nous com en la longitud dels mateixos. Per tant, la rebrotada induïda de l'arboç no ha de condicionar el fet d'aplicar o no la desbrossada del sotabosc, ja que la continuïtat vertical que es pugui desenvolupar entre rebrots nous i la capçada serà similar entre tractaments. Un any després de l'aplicació dels tractaments, l'arboç genera uns 28 rebrots nous de mitjana, mentre que s'han descrit xifres inferiors per *Q. cerrroides* i superiors per *Q. ilex* (14 i 35 respectivament, Espelta et al. 2003). Per tant, es podria pensar que l'arboç té un

comportament, en termes de dinàmica de creixement, intermedi entre les espècies rebrotadores caducifòlies i les perennifòlies, degut possiblement a la seva naturalesa de baixa esclerofília. Aquesta comparació suggereix que el grau de manteniment de les fulles podria estar relacionat amb el comportament de la rebrotada de les espècies arbòries.

Els tractaments forestals estudiats també produeixen un efecte positiu i a mitjà termini en el creixement del conjunt de la bosquina d'arboç. Així, els arboços de les parcel·les amb selecció de rebrots i desbrossada del sotabosc (considerant els individus als que s'havia aplicat els tres tractaments a nivell individual) van créixer més en la longitud dels seus rebrots reservats respecte els arboços de les parcel·les on es va aplicar només la selecció de rebrots; i aquests últims, a la vegada, van créixer més que els arboços de les parcel·les control (capítol 2). És a dir, que fins i tot els individus control ubicats a les parcel·les tractades, a mitjà termini també van créixer més en la longitud dels seus rebrots, que els individus control de les parcel·les control. Per tant, quan ens trobem en alzinars, rouredes, pinedes o qualsevol comunitat mediterrània on l'arboç sigui una espècie acompanyant, si es realitza un aprofitament forestal de l'espècie dominant i es desbrossa el sotabosc, encara que no es gestionin els individus d'arboç acompanyants, els resultats suggereixen que aquests també creixeran més en longitud degut a un efecte d'alliberament de recursos a nivell de la comunitat. Aquest efecte també es podria manifestar en altres espècies arbustives o arbòries acompanyants.

Per altra banda, trobem que la desbrossada del sotabosc no provoca una millora a curt termini del creixement de l'arboç, sinó que aquest es dona al cap de tres anys de l'aplicació del tractament. El creixement anual de la longitud dels rebrots reservats dels arboços seleccionats en parcel·les desbrossades és del doble que el dels arboços només seleccionats (6,1cm i 3,1cm, respectivament). En un context de recursos econòmics limitats en el sector forestal, cal valorar si val la pena destinar una partida per a realitzar la desbrossada del sotabosc (entre 2.580 a 7.140 €/ha segons el pendent) (Diputació de Barcelona 2013), ja que per una banda, es redueix el risc d'incendi i es millora el creixement a mitjà termini dels arboços, però per l'altra, amb l'aplicació de només la selecció de rebrots d'arboç ja es genera, a curt termini, una millora del seu creixement en longitud i en diàmetre, així com una reducció del risc d'incendi al disminuir part de la continuïtat horitzontal de combustible.

L'aplicació d'ambdós tractaments estimula el creixement vertical en el conjunt de la bosquina, però alhora, també redueix la quantitat de combustible mort en peu (34% de rebrots morts en els individus control, enfront de l'1% de rebrots morts en els individus tractats; capítol 2). Per tant, aquests tractaments permeten millorar l'estructura de les

bosquines d'arboç i reduir-ne el seu risc d'incendi, formant un paisatge més favorable davant l'actual escenari d'incendis forestals a la Conca Mediterrània.

2.2. Cicle fenològic i millora de la qualitat dels fruits

Hi ha bibliografia que recull informació contradictòria sobre la cronologia de les fenofases del cicle reproductiu de l'arboç (Castro-Díez et al. 2003, Ogaya i Peñuelas 2007), probablement degut a la complexitat de les fases del seu cicle vital.

Amb aquest treball, s'ha analitzat el cicle vital de l'arboç de la població estudiada amb molt detall i esforç de mostreig (Figura 1, capítol 3). Els nostres resultats mostren que, bàsicament, durant l'estiu es comencen a formar les estructures de les flors, de manera que al setembre ja s'inicia la formació de les mateixes, les quals es desenvolupen totalment a mitjans de novembre. Les flors són pol·linitzades a l'hivern i immediatament apareixen els fruits verds, els quals van agafant, inicialment, una tonalitat vermellosa. Els fruits verds van incrementant el seu diàmetre, el qual esdevé màxim sobre el mes de juny i la seva tonalitat esdevé groc-taronja. Els fruits assolixen el pic de màxima maduració durant la segona quinzena d'octubre. Una de les característiques reproductives de l'arboç és que es poden observar flors i fruits madurs en la mateixa època de l'any.

Els tractaments silvícoles aplicats produeixen un major número de fruits madurs per individu i alhora, aquests tenen un major pes sec i una major proporció de llavors desenvolupades (capítol 3), implicant una millora de la seva qualitat. Aquest resultat implica que l'alliberament de recursos degut a l'aplicació dels tractaments no només és emprat per estimular el creixement de l'arboç, sinó també per millorar la seva reproducció, com s'ha mostrat amb altres espècies (Levins 1968, Obeso 2002, Sánchez – Humanes i Espelta 2011), de manera que no es produeix un trade-off negatiu entre creixement i reproducció.

La disponibilitat d'una major quantitat de fruits madurs per individu pot ser d'especial rellevància a nivell de la bosquina, per la regeneració de la població. Malgrat que els tractaments redueixen el nombre de rebrots per hectàrea, s'aconsegueix mantenir la densitat de llavors desenvolupades per hectàrea, garantint el potencial reproductiu de la població. Alhora, uns fruits més atractius pel seu pes sec superior (Debussche i Isenmann 1989; Herrera et al. 1994 per *Phillyrea latifolia*), poden ser més depredats i dispersats per part de la fauna, afavorint la germinació i l'establiment de noves plàntules d'arboç.



Figura 1. Exemple del cicle reproductiu de l'arboç: a. estructures florals; b. formació de les flors; c. flors completament formades; d. fruits verds; e i f. fruits verds de major diàmetre; g. fruits mig madurs; h. fruits madurs; i. flors i fruits madurs; j. fruits madurs picats per ocells; k. fruits madurs caiguts a terra.

L'arboç es caracteritza per ser una espècie amb poca capacitat d'establiment de noves plàntules (taxa d'establiment d'unes 13-15 plàntules/ha i any; Lídia Quevedo, observació personal; Arianoutsou i Diamantopoulos 1985), establint-se, preferentment en zones de vegetació densa (Mesléard i Lepard 1991). Per tant, caldria dur a terme estudis addicionals que permetin avaluar aquesta possible millora en la reproducció sexual de la població i la seva capacitat de colonització de nous espais.

2.3. Resposta de la comunitat faunística

Els aprofitaments forestals arreu del món poden tenir efectes positius, negatius o bé neutres sobre la biodiversitat animal dels ecosistemes gestionats (Thiollay 1997, Whitman et al. 1998, Vasconcelos et al. 2000, Davis et al. 2001, Lewis 2001, Sverdrup-Thygeson i Ims 2002, Dunn 2004, Gómez i Abril 2011). En el nostre cas, els tractaments forestals aplicats es podrien entendre com una segona pertorbació després del foc, que pot afectar la diversitat de diferents grups d'organismes (Moya et al. 2009). Els nostres resultats demostren que aquests tractaments silvícoles no impliquen cap alteració sobre l'estructura i composició de la comunitat de formigues. Probablement, aquest nul impacte és degut a que la reducció de rebrots i de l'estrat arbustiú no són prou importants com per modificar les condicions de l'hàbitat de les formigues, així com les seves font d'aliment. A partir d'aquests resultats i considerant que les comunitats de formigues poden ser bones indicadores de la biodiversitat (Vasconcelos 1999a, Hoffmann i Andersen 2003, Nakamura et al. 2007, Andersen et al. 2009, Gómez i Abril 2011) podem considerar que són tractaments poc agressius per la diversitat de la fauna invertebrada. Tot i així, seria interessant plantejar altres estudis que avaluin l'efecte d'aquests tractaments a una escala diferent a l'estudiada (formigues com a bioindicadores dels invertebrats), és a dir, sobre altres tipus de comunitats animals que es podrien veure afectats per aquests tractaments, ja sigui positivament o negativa, com per exemple, les aus o els micromamífers.

L'augment de la quantitat de fruits madurs per individu d'arboç, així com la millora de la qualitat dels mateixos, derivats de l'aplicació dels tractaments forestals, podria beneficiar la fauna que s'alimenta de l'arboç, ja que aquests tractaments garanteixen la quantitat d'aliment disponible, és a dir, del número de flors i de fruits madurs per hectàrea. Això és rellevant, per un costat, per espècies que s'alimenten de nèctar i pol·len de les seves flors, com abelles i abellots (*Apis mellifera*, *Bombus* sp.; Herrera 2004, Rasmont et al. 2005), ja que, a més, l'arboç produeix les flors a la tardor, una època de l'any en què hi ha poques espècies florides al Mediterrani (Picó i Retana

2001). Per un altre costat, en ser l'espècie més abundant, és potencialment la font d'aliment principal per a la fauna que s'alimenta de fruits carnosos, amb un important contingut en carbohidrats i sucres (Ayaz et al. 2000, Barros et al. 2010); o bé de llavors, com aus, mamífers i formigues (Herrera 1982, Arianoutsou i Diamantopoulos 1985, Christianini i Oliveira 2010, Virgós et al. 2010, Rost 2011), especialment en una època de l'any desfavorable per a la fauna (tardor-hivern) per l'escassa presència d'altres fruits. Així, una major quantitat i qualitat de fruits d'arboç disponible podria millorar la situació alimentària de la comunitat faunística, assegurant la seva supervivència.

Un punt en el que cal fer especial menció i que, tot i no estar considerat en el plantejament inicial de la tesi, ha estat un resultat d'interès, és l'aparició de la plaga de l'eruga *Euproctis chrysorrhoea* L.. Aquesta espècie pot aparèixer i fer explosions demogràfiques que afectin a bona part de les bosquines d'arboç, ja que s'alimenten dels brots tendres d'aquesta espècie, els quals són especialment abundants en els arboços tractats. Tot i així, hem pogut constatar que malgrat l'eruga té una major presència en aquelles parcel·les amb els arboços tractats amb selecció de rebrots i desbrossada del sotabosc, els resultats mostren que el possible efecte negatiu que generen queda contrarestat amb l'efecte estimulants en el creixement i la reproducció que generen els tractaments. De fet, els individus d'arboç tractats, amb més presència d'erugues, creixen més en longitud i diàmetre i produeixen fruits de major qualitat que els individus no tractats. Per tant, no ha de fer por plantejar l'aplicació de tractaments de millora forestal en bosquines d'arboç si aquestes compten amb la presència de l'eruga *Euproctis chrysorrhoea* L., o si hi ha la possibilitat que siguin infectades un cop aplicats els tractaments. Alhora, és probable que aquesta explosió demogràfica duri pocs anys, tal com s'ha observat en la nostra zona d'estudi. No obstant, cal valorar la possibilitat de prevenir aquesta plaga sempre que sigui possible, a través de controls biològics, com per exemple, l'ús d'olis essencials d'anís (*Pimpinella anisum*), d'orengues (*Origanum onites* i *O. minutiflorum*), de nematodes i parasitoids (Erler i Cetin 2008, 2009; Nikdel et al. 2010, Frago et al. 2012), ja que sense ella els efectes dels tractaments silvícoles sobre l'arboç possiblement encara seran més positius.

3. Consideracions generals per a la gestió de les bosquines d'arboç

Amb aquest treball hem aconseguit disposar d'una visió integrada de la resposta de les bosquines d'arboç a l'aplicació d'uns tractaments silvícoles postincendi, de manera

que integrem diferents aspectes relacionats directament amb l'arboç: en les seves respostes en creixement i reproducció, així com l'efecte sobre la fauna associada. Amb aquests tractaments hem comprovat que s'estimula el creixement dels arboços i es millora la qualitat dels seus fruits, sense alterar la diversitat de formigues, a la vegada que es redueix el risc d'incendi. Amb aquesta gestió es podria potenciar l'aprofitament dels seus recursos, com la fusta i llenya (Galán et al. 1998), la producció de mermelades i licors, fins i tot per extreure components per determinats medicaments (Kivcak et al. 2001, Gomes i Canhoto 2009, Molina et al. 2011). Per tant, les cada vegada més presents bosquines d'arboç en el nostre territori poden ser gestionades satisfactòriament per obtenir-ne un aprofitament dels recursos derivats i, a la vegada, millorar la prevenció d'incendis forestals.

Tanmateix, cal dur a terme futurs treballs més a mitjà i llarg termini per avaluar la resposta d'aquestes bosquines d'arboç en una escala temporal més llarga i valorar si realitzar una selecció de rebrots cada certs anys pot millorar significativament aquestes bosquines. En cas afirmatiu, caldria determinar quina freqüència seria necessària per millorar l'estructura d'aquestes cobertes, així com el seu impacte sobre la biodiversitat, incloent altres grups d'organismes que es poden veure afectats.

Aquest treball presenta un marc conceptual i experimental que pot ser útil per la gestió de bosquines d'espècies mediterrànies amb un gran vigor de rebrotada i un port arbustiu mitjà-alt. En l'actual context d'una elevada recurrència de focs a la Conca Mediterrània, poden emergir bosquines dominades per altres espècies rebrotadores, com ara, *Phillyrea angustifolia* o *Pistacia lentiscus* (Sabaté et al. 1990, Manes et al. 2002, Quevedo et al. 2007, Vitale et al. 2007). Davant d'aquest nou tipus de cobertes, caldrà definir quins tractaments forestals són els més adequats per obtenir una bona estructura, creixement i reproducció de l'espècie, amb la finalitat que els propietaris forestals puguin realitzar algun tipus d'aprofitament, reduint el risc d'incendi i minimitzant l'impacte sobre la biodiversitat.

De la mateixa manera que s'ha realitzat amb l'arboç, la realització de treballs amb altres espècies rebrotadores que poden arribar a dominar algunes formacions forestals, pot ajudar a treure unes conclusions globals on, a partir de les característiques funcionals de les espècies, puguem predir quines seran les seves respostes davant d'una gestió forestal. D'aquesta manera es podrien elaborar uns models globals de gestió, els quals poden esdevenir una eina pràctica i útil per als gestors forestals.

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