Universitat de Girona

L'EFECTE DE LA GESTIÓ FORESTAL POST-INCENDI SOBRE L'AVIFAUNA I LA DISPERSIÓ DE LLAVORS PER OCELLS FRUGÍVORS

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Dipòsit legal: GI-1025-2011
http://hdl.handle.net/10803/32137

Universitat de Girona

TESI DOCTORAL

L'efecte de la gestió forestal post-incendi sobre l'avifauna i la dispersió de llavors per ocells frugívors

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2011

PROGRAMA DE DOCTORAT
EN CIÈNCIES EXPERIMENTALS I SOSTENIBILITAT

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Memòria presentada per optar al títol de doctor per la Universitat de Girona

Universitat de **Girona**

El Dr. Pere Pons Ferran i el Dr. Josep Maria Bas Lay, del Departament de Ciències Ambientals de la Universitat de Girona,

CERTIFIQUEN:

Que aquest treball, titulat "L'efecte de la gestió forestal post-incendi sobre l'avifauna i la dispersió de llavors per ocells frugívors", que presenta Josep Rost Bagudanch per a l'obtenció del títol de doctor, ha estat realitzat sota la seva direcció.

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Girona, abril de 2011.

Agraïments

Per molt que a la portada del llibret que teniu entre mans només hi surt un nom i uns cognoms, la veritat és que en aquesta tesi hi ha participat molta gent, en menor o major mesura. Tots aquells que m'he trobat al llarg d'aquests anys de doctorat heu contribuït d'alguna manera a aquest treball. De manera que, companys i companyes, us haig d'agraïr a tots els que m'he trobat en aquest temps la vostra paciència, la vostra companyia, els vostres ànims, les vostres converses, les vostres crítiques, els vostres cops de mà. El fet que aquesta etapa hagi resultat una experiència fantàstica també ha estat cosa vostra! Comença doncs, l'apartat on em deixen posar la resta d'autors de la tesi. Espero no deixar-me'n cap!

Haig de reconèixer, com diuen els sicilians, que el primer és la família. El suport que m'han donat sempre els meus **pares** i la meva **germana** ha estat incondicional, no se us pot demanar res més! Vaja... excepte si pot ser, de tant en tant algun tàper el diumenge, que la paella encara no la tinc controlada del tot!

I ja mirant més cap a la universitat, al primer a qui haig de donar un gràcies inacabable és a en **Pere**. La veritat és que no podia haver tingut un director de tesi millor, i ho dic de la manera més sincera. Un director fantàstic, que m'ha fet treballar i amb qui he treballat molt bé, i que sempre ha estat allà al costat. Exactament, a dos despatxos de distància. De fet, al mateix despatx que en **Pitu**, al que li haig d'agrair la codirecció i l'ajuda al camp, sobretot al començament. I a ells dos conjuntament els hi haig de donar les gràcies de donar-me l'oportunitat de treballar quatre anys en uns llocs immillorables (Mèxic, Montana, l'Empordà... qui pot demanar més?), i diria que sobretot, amb uns companys immillorables: els del despatx, en **Josep**, la **Núria** i el gran **Cape**, els de cafès a mig matí i tertúlia, en **David**, l'**Albert**, i en **Miguel**, que es mereix comentari apart perquè ha estat un autèntic codirector, fonamental en moltes parts de la tesi, a més de per fer-nos passar sempre un bon rato amb la seva alegría naturá. I a la resta de l'àrea també gràcies per les bones estones compartint reunions, dinars i alguna costellada: la **Sílvia A.**, la **Malú**, la **Gabi**, la **Marta**, la **Margarida**, en **Cris**, la **Caterina** (i pel peix!), la **Dolors**, l'**Ahmed**, la **Sílvia V.**, l'**Eulàlia** i la **Mireia**.

També haig d'agrair als companys de projecte de Solsona les facilitats que m'han posat sempre per treballar amb ells, en especial en **Lluís** per aportar el seu sempre interessant punt de vista en alguns dels treballs de la tesi, i a l'**Elena** i en **Vicent** per la gestió tan eficient de les dades del DINDIS.

També haig de donar les gràcies a en **David Meya**, del Departament, que m'ajudés a entendre una mica més la gestió de boscos cremats a Catalunya i els seus condicionants econòmics, informació que ha estat fonamental en un treball com aquest.

També es mereixen que els hi doni les gràcies tots els companys dels àpats de tàper a la facultat. El dinar i el cafè de després no s'haurien pas posat tan bé sense la companyia d'en **Joan N.**, la **Natàlia**, la **Berta**, la **Susanna**, en **Lorenzo**, en **Jordi**, la **Meri**, la

Publicacions derivades de la tesi doctoral

La tesi doctoral que es presenta té un format de tesi com a compendi d'articles científics. Els cinc capítols de què consta han estat redactat com a tals i en el moment de presentació de la tesi es troben o bé publicats o bé en procés de revisió en revistes indexades al Science Citation Index. A continuació es detalla en quin punt del procés editorial es troba cadascun i s'aporten indicis que justifiquen que aquests treballs tenen la qualitat necessària per a la presentació de la tesi doctoral en aquest format. Els factors d'impacte (FI) de les publicacions corresponen a la darrera actualització, de l'any 2009.

1. Rost, J., Clavero, M., Brotons, L. & Pons, P. The effect of postfire salvage logging on bird communities in Mediterranean pine forests: the benefits for declining species. Journal of Applied Ecology (en segona revisió).

   FI: 4,197. Posició 20/129 (Q1) de la categoria *Ecology*.

2. Rost, J., Hutto, R.L., Brotons, L. & Pons, P. Comparing the effect of salvage logging on birds in the Mediterranean basin and the Rocky Mountains: common patterns, different management implications. Conservation Biology (en revisió).

   FI: 4,666. Posició 15/129 (Q1) de la categoria *Ecology*, 2/92 (Q1) en la categoría *Biodiversity Conservation* i 7/181 (Q1) de la categoría *Environmental Sciences*.

3. Rost, J., Clavero, M., Bas, J.M. & Pons. P. 2010. Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests. Forest Ecology and Management, 260: 79-86.

   FI: 1,950. Posició 5/46 (Q1) a la categoria *Forestry*.

4. Rost, J., Pons, P. & Bas, J.M. 2009. Can salvage logging affect seed dispersal by birds into burned forests? Acta Oecologica 35, 763-768.

   FI: 1,609. Posició 66/129 (Q3) a la categoria *Ecology*.

5. Rost, J., Bas, J.M. & Pons, P. The importance of piling wood debris on the distribution of bird-dispersed plants in burned and logged Mediterranean pine forests. International Journal of Wildland Fire (acceptat).

   FI: 1,901. Posició 6/46 (Q1) a la categoria *Forestry*.

ÍNDEX

Capítol 3. *Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests.*

Capítol 4. *Can salvage logging affect seed dispersal by birds into burned forests?*

Capítol 5. *The importance of piling wood debris on the distribution of bird-dispersed plants in burned and logged Mediterranean pine forests.*

Discussió general i conclusions

SUMMARY

After a wildfire, a second severe disturbance usually takes place in conifer forests: salvage logging. This kind of management is carried out basically for an economic purpose, and involves a drastic change in the habitat structure. The short-term effect of these two disturbances on birds is analyzed in this thesis, with different objectives. First of all, the response of the different bird species to postfire logging is analyzed during the three years after fire, focusing especially on those of conservation interest. Considering that in Europe a number of open-habitat birds hold an unfavorable conservation status, salvage logging may favor their presence after the removal of snags. On the other hand, the response of birds to salvage logging is compared between the Mediterranean Basin (Catalonia) and the North-American Rocky Mountains (Montana), where conservation priorities are different and fall on some fire specialist species. A second aim is to assess the change in the bird community in the breeding season, which was studied trough variables such as species richness, density, species composition and the overall community conservation value. Finally, the effect of postfire salvage logging is assessed on ecosystem processes in which birds are involved. Seed dispersal of fleshy-fruited plants that several frugivorous birds carry out is one of these processes. Specifically, the influence of salvage logging on the frugivores' habitat preferences, seed rain and the distribution of dispersed plants is studied. The results show that postfire salvage logging determines very significantly the presence of most species in burned Aleppo pine forests, being detrimental for forests birds and facilitating the colonization of the area by open-habitat birds. As a result of such species turnover, which takes place without an overall loss in richness and density values (despite existing important interannual variations), the overall community conservation value increases. However, there are also species of conservation interest that are affected negatively by salvage logging. Seed dispersal by birds is also altered by postfire management. Seed rain decreases in general, exceptuating wood debris or slash piles built using logging remnants, where seed dispersal was comparable to what can be found in adjacent unburned pine forests. These structures also enhance the presence of frugivorous species in winter, one of the most important seasons for fleshy-fruited plants' seed dispersal in the Mediterranean region. These plants were found in higher abundances within these wood debris piles than outside of them, particularly in dry areas. Therefore, postfire salvage logging has a very important influence on birds and on ecosystem process in which birds are involved. A heterogeneous postfire management that combined logged areas (leaving a minimal snag density) with non-intervention areas would be probably the most suitable for both open-habitat and other species that are also of conservation interest but require the presence of snags, as well as for those bird species that play an important role in the seed dispersal process.

RESUMEN

Después de un incendio forestal, en bosques de coníferas suele tener lugar una segunda perturbación severa: la tala de los árboles quemados. Este tipo de gestión responde principalmente a motivos económicos, y tiene como consecuencia un cambio drástico en la estructura del hàbitat. En esta tesis se pretende analizar el efecto combinado de estas dos perturbaciones sobre la avifauna, a corto plazo y con diferentes objetivos. Inicialmente, se estudia qué especies se ven más afectadas por la tala durante los tres primeros años después del fuego, poniendo énfasis en aquellas que tienen interés de conservación. Considerando que en Europa muchas aves de zonas abiertas están en situación desfavorable, la tala podría favorecer su presencia por la eliminación de los árboles quemados. Por otra parte, se compara la respuesta de las aves a la tala post-incendio entre la Cuenca Mediterránea (Cataluña) y las Montañas Rocosas norteamericanas (Montana), donde las prioridades de conservación son distintas y recaen principalmente en aves especialistas de bosques quemados. En un segundo objetivo, se estudia también cómo cambia la comunidad de aves nidificantes después de la tala, a través de la riqueza, la densidad, la composición de especies, y el valor de conservación de toda la comunidad. Finalmente, también se valora si la afectación de la tala post-incendio sobre la avifauna tiene efectos sobre procesos a nivel de ecosistema en los cuales participan las aves. El proceso de dispersión de semillas de plantas con fruto carnoso que llevan a cabo algunos frugívoros es uno de estos casos. Concretamente, se estudia cómo la gestión forestal puede alterar las preferencias de hàbitat de las aves frugívoras, la lluvia de semillas y la distribución de las plantas dispersadas. Los resultados de los diferentes capítulos muestran como la tala determina de un modo muy importante la presencia de la mayoría de especies en bosques de pino carrasco quemados, perjudicando las forestales y facilitando la colonización de las que son propias de zonas abiertas. Como resultado de esta sustitución de especies, que se produce sin pérdidas globales en la riqueza y la densidad (aunque con variaciones interanuales considerables), el valor de conservación de la comunidad aumenta después de la tala. Pese a esto, algunas aves con interés de conservación también se ven perjudicadas por la tala. El proceso de dispersión de semillas también se ve afectado por la gestión post-incendio. La lluvia de semillas disminuye en general, excepto en los montones de ramas construidos con restos de árboles cortados, donde la dispersión fue equiparable a la encontrada en pinares próximos no quemados. Estas mismas estructuras favorecen la presencia de frugívoros en invierno, una de las épocas más importantes para la dispersión de semillas de plantas con fruto carnoso en la región mediterránea. Estas plantas se encontraron en abundancias más elevadas en estos montones de ramas que fuera de ellos, especialmente en las zonas más secas. La tala post-incendio, por lo tanto, tiene un efecto muy importante sobre la avifauna y sobre procesos a nivel de ecosistema en los que las aves están implicadas. Una gestión heterogénea, alternando áreas cortadas (dejando una mínima densidad de pies) con otras sin intervención probablemente sea lo más adecuado para favorecer tanto especies de zonas abiertas como otras que también tienen interés de conservación y que requieren la presencia de árboles, y también las que desempeñan un papel importante en la dispersión de semillas.

RESUM

Després d'un incendi forestal, als boscos de coníferes acostuma a tenir lloc una segona pertorbació severa: la tala dels arbres cremats. Aquest tipus de gestió respon principalment a motivacions econòmiques, i té com a conseqüència un canvi dràstic en l'estructura de l'hàbitat. En aquesta tesi s'analitza l'efecte combinat d'ambdues pertorbacions sobre l'avifauna, a curt terme i amb diferents objectius. Inicialment, s'analitza quines espècies es veuen més afectades per la tala durant els tres anys posteriors al foc, posant un èmfasi especial en aquelles amb interès de conservació. Tenint en compte que a Europa moltes espècies d'espais oberts estan en una situació desfavorable, la tala podria afavorir la seva presència per l'eliminació dels arbres cremats. D'altra banda, es compara la resposta dels ocells a la tala post-incendi entre la Conca Mediterrània (Catalunya) i les Muntanyes Rocalloses nord-americanes (Montana), on les prioritats de conservació són diferents i recauen bàsicament en especialistes de boscos cremats. En un segon objectiu, s'estudia com canvia la comunitat d'ocells nidificants després de la tala, a través de la riquesa, la densitat, la composició d'espècies i el valor de conservació de tota la comunitat. Finalment, també es valora si l'afectació de la tala post-incendi sobre l'avifauna té efectes sobre processos a nivell d'ecosistema en els quals participen els ocells. El procés de dispersió de llavors de plantes amb fruit carnós que duen a terme alguns frugívors és un d'aquests casos. En concret, s'estudia de quina manera la gestió altera les preferències d'hàbitat dels ocells frugívors, la pluja de llavors i la distribució de les plantes dispersades als boscos cremats gestionats. Els resultats dels diferents capítols mostren com la tala determina de manera molt important la presència de la majoria d'espècies en boscos de pi blanc cremats, perjudicant els forestals i facilitant la colonització dels d'espais oberts. Com a resultat d'aquesta substitució d'espècies, que es produeix sense pèrdues globals en la riquesa d'espècies i la densitat (malgrat haver-hi variacions interanuals considerables), el valor de conservació de la comunitat augmenta després de la tala. Tot i això, algunes espècies amb interès de conservació també es veuen perjudicades per la tala. El procés de dispersió de llavors per ocells també es veu afectat per la gestió post-incendi. La pluja de llavors disminueix en general, excepte als pilons de branques o feixines construïts amb restes dels arbres tallats, on la dispersió va ser equiparable a la trobada en pinedes no cremades properes. Aquestes mateixes estructures afavoreixen també la presència de frugívors a l'hivern, una de les èpoques més importants per la dispersió de llavors de plantes amb fruit carnós a la regió mediterrània. Aquestes plantes es van trobar en abundàncies més elevades en aquests pilons de branques que fora dels mateixos, especialment en vessants més secs. La tala post-incendi, per tant, té un efecte molt important sobre l'avifauna i sobre processos a nivell d'ecosistema en els que estan implicats els ocells. Una gestió heterogènia alternant àrees tallades (deixant una mínima densitat de peus) amb altres sense intervenir probablement sigui el més adient per afavorir tant espècies d'espais oberts com d'altres que també tenen interès de conservació i que requereixen la presència d'arbres, així com aquelles espècies que tenen un paper important en la dispersió de llavors.

Introducció general

INTRODUCCIÓ GENERAL

Foc, fauna i gestió post-incendi

El foc és una de les pertorbacions ecològiques que incideix de manera més important sobre els ecosistemes forestals de tot el món, tant des d'un punt de vista ecològic com també econòmic (Blondel & Aronson 1999, Hessburg & Agee 2003, FAO 2007). Com a pertorbació natural, forma part de la pròpia dinàmica d'aquests ecosistemes, les comunitats vegetals i faunístiques dels quals han conviscut i evolucionat amb el foc durant mil·lennis, i hi estan adaptades o fins i tot en depenen (Hutto 2008, Baker 2009, Pausas & Keeley 2009). Tanmateix, la percepció que la societat té dels incendis actualment és que es tracta de fenòmens catastròfics que cal evitar i contra els quals cal lluitar, més que no pas de processos naturals. Aquesta visió negativa del foc, juntament amb les pèrdues econòmiques i, en alguns casos, de vides humanes, ha portat a promoure'n la supressió en algunes àrees i a augmentar els esforços en tasques de prevenció i, sobretot, d'extinció d'incendis (Beschta *et al.* 2004; Lindenmayer *et al.* 2004).

Paradoxalment, però, en les darreres dècades el nombre i l'extensió dels focs forestals han augmentat de manera molt significativa en zones on aquestes polítiques s'han dut a terme (Keane *et al.* 2002, Plana 2004, Pausas *et al.* 2008). Diversos factors poden haver jugat un paper important en aquest fet. En zones de Nord Amèrica, les polítiques de supressió del foc s'han traduït en canvis importants en el paisatge, on ha augmentat de manera significativa la superfície i la continuïtat dels boscos en detriment d'ambients més oberts, i de gran interès, com les praderies o els matollars (Keane 2002, Hessburg & Agee 2003). A la regió mediterrània, en dècades recents els canvis dels usos del sòl han tingut un efecte similar, atès que l'abandonament de les activitats agrícoles ha donat pas també a l'augment de la superfície forestal (Moreira *et al.* 2001, Pausas *et al.* 2008). Aquests canvis, entre altres coses, han permès que tant la quantitat de combustible com la continuïtat de superfície forestal augmentés de forma important, propiciant unes condicions adients per a l'ocurrència de grans incendis forestals (Plana 2004). A més a més, el risc d'ignició està molt relacionat amb la presència humana, de manera que l'increment en les últimes dècades del nombre d'urbanitzacions, segones residències, presència de línies de conducció elèctrica, etc. en zones d'interfase entre els ambients urbà i rural-forestal, també ha contribuït a augmentar el nombre d'incendis en aquestes zones (Plana 2004, Pausas *et al.* 2008). D'altra banda, les prediccions sobre la incidència dels incendis forestals apunten que en les properes dècades el seu nombre s'incrementarà, tant a nivell global com a la regió mediterrània (Piñol *et al.* 1998, Krawchuk *et al.* 2009), de manera que la seva importància en la configuració del paisatge mediterrani probablement augmentarà en el futur.

Els incendis suposen una pertorbació important en l'ecosistema forestal, la magnitud del qual depèn tant de les característiques del propi foc com de les del bosc que es crema. La resposta de les comunitats faunístiques al foc ha estat estudiada en diferents grups (e.g. cargols: Santos *et al.* 2009; formigues: Arnan *et al.* 2006, Rodrigo & Retana 2006; coleòpters: Kaynaş & Gürkan 2005, Rodrigo *et al.* 2008, Fattorini 2010; rèptils: Lindenmayer *et al.* 2008, Santos & Poquet 2010; mamífers: Fisher & Wilkinson 2005, Henriques *et al.* 2006, Cunningham *et al.* 2006), destacant però l'atenció dedicada als ocells en aquest camp, amb estudis duts a terme en diferents regions biogeogràfiques, a diferents escales i sobre espècies concretes o sobre comunitats senceres (e.g. Hutto 1995, Pons & Prodon 1996, Herrando *et al.* 2003, Barlow & Peres 2004, Brotons *et al.* 2005, Smucker *et al.* 2005, Hutto 2008, Lindenmayer *et al.* 2008, Vallecillo *et al.* 2008, Jacquet & Prodon 2009). No obstant, l'atenció que s'ha dedicat als ocells en ambients cremats presenta un biaix regional important, tenint en compte que més de la meitat dels estudis corresponen a treballs desenvolupats a Nord Amèrica (Leidoff & Bissonette 2009). Europa representa menys del 10% d'aquests treballs, i tot i que la majoria corresponen a la regió mediterrània (Prodon & Pons 1993), la informació disponible sobre aspectes concrets d'aquest tema, com l'efecte de la gestió post-incendi sobre la fauna, és certament limitada.

Després d'un foc forestal, sovint té lloc una segona pertorbació, que també implica un canvi important en la complexitat estructural de l'hàbitat (Figures 1 i 2). Aquest canvi addicional està directament provocat per l'home, i ve donat per la tala dels arbres cremats. La tala post-incendi és la mesura de gestió més corrent després que un bosc es cremi a moltes regions del món (Bautista *et al.* 2004, Beschta *et al.* 2004, Lindenmayer *et al.* 2004). La conveniència o no d'aquest tractament és un tema que ha estat objecte d'un intens debat en els darrers anys, pels efectes ecològics que pot tenir: a més de l'evident canvi en l'hàbitat cremat, també pot alterar cicles de nutrients i processos com l'erosió del sòl i la regeneració de la vegetació. Aquests efectes han fet que la seva aplicació s'hagi qüestionat en diverses ocasions des de la comunitat científica (McIver & Starr 2000, Lindenmayer *et al.* 2004, Beschta *et al.* 2004, Dellasala *et al.* 2006, Hutto 2006).

Motivacions de la tala post-incendi

La tala post-incendi s'ha convertit en el principal tipus de gestió de zones cremades a molts països de món a causa d'un conjunt de condicionants econòmics, silvícoles, de restauració, i fins i tot estètics. Algunes d'aquestes motivacions responen també a la visió del foc com una catàstrofe ecològica, i a la idea que cal rehabilitar el bosc cremat d'alguna manera per tornar a l'estat pre-incendi tan ràpidament com es pugui (Beschta *et al.* 2004, Lindenmayer *et al.* 2004, Schmiegelow *et al.* 2006). En qualsevol cas, la principal motivació d'aquest tipus de gestió sol ser l'econòmica. En explotacions forestals on l'interès principal és la comercialització de la fusta, retirar els arbres cremats permet recuperar part de les pèrdues ocasionades pel foc (Beschta *et al.* 2004,

Figura 1. Pineda de pi blanc *Pinus halepensis* cremada en l'incendi de Ventalló (entre l'Alt i el Baix Empordà) de l'agost de 2006. La fotografia va ser presa el mes de juny de l'any següent.

Figura 2. La major part de l'àrea cremada durant l'incendi de Sant Llorenç Savall (Vallès Occidental), l'agost de 2003, va ser tallada en els anys següents, donant com a resultat el paisatge que es mostra a la fotografia (presa el maig de 2009, sis anys després del foc).

Bautista *et al.* 2004, Castro *et al.* 2009). Tot i cremada, la fusta conserva part del seu valor; en el cas de Catalunya, per exemple, on la major part d'aquesta fusta és de pi (bàsicament pi blanc *Pinus halepensis* i pinassa *Pinus nigra*), es pot destinar a la fabricació de palets o taulons, o es pot triturar i obtenir-ne conglomerat de fusta i combustible de biomassa (Figures 3 i 4). L'extracció de la fusta cremada amb una finalitat comercial, tanmateix, s'ha de fer amb una relativa celeritat

(durant els primers anys després del foc) per no accentuar la pèrdua del seu valor. Tornant al cas de Catalunya, les feines de tala se solen dur a terme durant els dos primers anys, per evitar que la fusta es vegi massa afectada per l'activitat de coleòpters barrinadors o per fongs, la qual cosa empitjora la seva sortida al mercat. D'altra banda, la tala dels arbres també serveix per a facilitar l'accés a la zona afectada i per a dur a terme altres treballs silvícoles més endavant, atès que si no es retiren els troncs dels arbres morts que van caient amb el pas del temps, el trànsit dins l'àrea cremada es fa realment difícil. De tota manera, la tala d'arbres cremats es duu a terme també en altres boscos on la principal motivació no és econòmica, com en boscos públics o en àrees on la fusta no és rendible. En aquests altres casos, es proposen altres arguments a favor de la tala post-incendi, alguns dels quals han estat desenvolupats en la casuística dels boscos nord-americans i han estat després extrapolats com a vàlids a altres regions, com la Conca Mediterrània.

Un dels arguments que s'utilitzen per a justificar la tala de pins cremats (com també després d'altres pertorbacions severes) és que els arbres poden convertir-se en focus de coleòpters barrinadors (escolítids, buprèstids o cerambícids) que podrien esdevenir una plaga i afectar masses forestals veïnes no cremades (Amman & Ryan 1994). Tot i això, la majoria d'aquests barrinadors generalment només ataquen arbres vius debilitats, com aquells que s'han cremat parcialment, però no afecten arbres morts completament cremats –que són la majoria d'arbres que es poden trobar després dels focs de capçada que solen tenir lloc a la regió mediterrània (Eckberg *et al.* 1994, McCullogh *et al.* 1998, Bautista *et al.* 2004, Martikainen *et al.* 2006). La tala d'arbres també s'ha proposat com un mètode per reduir el risc de nous incendis a la mateixa àrea, adduint que els troncs i branques cremats poden ser un combustible important per a futurs incendis. Aquest argument, tanmateix, s'ha posat en qüestió tenint en compte el baix poder d'ignició de la fusta que ja ha estat cremada, especialment els troncs més gruixuts (Castro *et al.* 2009). Fins i tot s'ha vist que la severitat del foc pot ser fins i tot més elevada en àrees on s'havien extret aquestes restes que en altres on no s'havien retirat (Kulakowski & Veblen 2007, Thomson *et al.* 2007). En altres ocasions la tala post-incendi es duu a terme amb l'objectiu de reduir l'erosió del sòl, que pot ser un problema important en determinades ocasions (Bautista *et al.* 2004, Pausas *et al.* 2008). Una part de la fusta talada (o tota), a partir dels troncs i/o les branques, es pot destinar a la construcció de barreres en pendents pronunciats amb l'objectiu de disminuir l'escorrentia, cosa que també es pot intentar amb la trituració i escampada de les restes cremades (McIver & Starr 2000, Bautista *et al.* 2004).

Figura 3. Tala post-incendi al foc de Ventalló (de l'agost 2006). En aquest cas, s'aprofitava només la fusta dels troncs més gruixuts, i les branques i restes més primes es van deixar sobre el terreny, escampades o en petits pilons. Fotografia presa el juny de 2009.

Figura 4. Trituració dels pins cremats a Ventalló. Aquí els arbres eren triturats completament al mateix lloc de tala, per obtenir-ne combustible de biomassa. Fotografia feta l'abril de 2008.

A l'estat espanyol, la necessitat de la gestió post-incendi fins i tot queda recollida a nivell legislatiu. A la Ley de Montes (BOE núm. 102 de 29 d'abril de 2006), s'hi indica "l'obligació de restauració dels terrenys incendiats", i es defineix el foc literalment com un "desastre ecològic". Aquesta obligació no implica explícitament la tala post-incendi, tot i que sí que s'emplaça les comunitats autònomes a "fixar les mesures encaminades a la retirada de la fusta cremada i a la restauració de la coberta vegetal afectada pels incendis". Això fa que en determinats casos, aquestes mesures fixades per les comunitats autònomes es tradueixin en forma de subvencions i ajuts per als propietaris dels boscos cremats. En el cas de Catalunya, quan la fusta cremada no és rendible econòmicament, s'obren sovint línees d'ajuts perquè el propietari gestioni (tali) aquestes parcel·les no comercials per tal de retirar els arbres que poden obstaculitzar futurs treballs silvícoles i també per "prevenir incendis, assolir la regeneració natural o artificial, eliminar els efectes de l'erosió del sòl, disminuir les possibles fonts de malalties per a masses forestals no afectades pel foc i esmorteir els impactes negatius sobre el paisatge" (resolució MAH/965/2006). Per acollir-se a aquesta convocatòria concreta, pels incendis de l'estiu de 2005, la gestió a realitzar implicava la tala dels arbres i arbustos cremats i que les restes fossin trossejades i escampades sobre el terreny; treballs pels quals els propietaris podien percebre fins a 1.150 euros/ha, que podien ser més en el cas que s'haguessin de construir o arranjar camins o pistes forestals (Generalitat de Catalunya 2006). A Catalunya, una altra opció per al propietari és la cessió dels terrenys a l'administració perquè gestioni la zona afectada segons un pla de gestió concret, amb garanties d'aconseguir una parcel·la explotable en un futur (D. Meya com. pers.).

Conseqüències de la tala post-incendi sobre la fauna

La tala post-incendi suposa una segona pertorbació sobre els ecosistemes forestals afectats pel foc, en tant que suposa una alteració important de la complexitat estructural de l'hàbitat, la qual cosa té un efecte sobre la fauna present en boscos cremats (McIver & Starr 2000, Hutto 2006). Després de pertorbacions severes, les estructures que formaven part de l'hàbitat i que es mantenen (els anomenats llegats biològics, o *biological legacies* en anglès) poden tenir una funció important per a algunes espècies i per a la recuperació de l'ecosistema, a través dels serveis ecològics que aquestes realitzen (Hutto 2006, Lindenmayer & Noss 2006). En el cas dels incendis, l'arbrat cremat en seria un exemple, atès que suposa un recurs important per a certes espècies que l'utilitzen com a lloc d'alimentació o reproducció (ja sigui a les capçades o en cavitats), de manera que la tala té com a resultat que algunes espècies que ocupaven el bosc cremat en desapareguin, com és el cas d'una bona quantitat d'ocells de medis forestals (Llimona *et al.* 1993, Izhaki & Adar 1997, Morissette *et al.* 2002, Cahall & Hayes 2009, Moneglia *et al.* 2009, Castro *et al.* 2010). En el cas dels ocells, a Nord Amèrica fins i tot existeixen picots (e.g. *Picoides arcticus*) que es consideren especialistes de boscos cremats d'elevada severitat, atès que seleccionen positivament

aquests ambients (Saab & Dudley 1998, Imbeau *et al.* 1999, Hoyt & Hannon 2002, Kotliar *et al.* 2002, Hutto & Gallo 2006, Hutto 2008).

No obstant això, algunes altres espècies d'ocells més lligades a ambients oberts o a estadis successionals inicials, colonitzen o esdevenen més abundants en llocs on s'han retirat els arbres que no pas en boscos cremats que s'han mantingut intactes (Izhaki & Adar 1997, Morissette *et al.* 2002, Castro *et al.* 2010). En el cas que els arbres no s'hagin eliminat completament (tales parcials o selectives), algunes espècies que crien en cavitats d'arbres, però que necessiten ambients més oberts que els forestals, poden ocupar aquestes zones (Kotliar *et al.* 2002, Saab *et al.* 2007). D'altra banda, la manera com es gestionen les restes de fusta morta no aprofitables comercialment (les branques més primes) també és important per a la fauna. Deixar les branques in situ, ja sigui escampades o en petits pilons, afavoreix la presència de més espècies d'ocells en època reproductora (Herrando *et al.* 2009). A banda dels ocells, la presència de brancam apilonat afavoreix la presència de conills (*Oryctolagus cuniculus*), una espècie clau dels ecosistemes mediterranis (Rollán & Real 2010), i també de micromamífers (Haim & Izhaki 1994, Loeb 1999, Tiedemann *et al.* 2000, Manning & Edge 2005) i d'algunes espècies de rèptils (Santos & Poquet 2010).

Aquests canvis en la presència i abundància de determinades espècies també poden tenir, en conseqüència, un efecte en els processos de l'ecosistema en els quals aquestes espècies participen. Per exemple, la dispersió de llavors de plantes amb fruits carnosos és un procés clau per al manteniment de bones poblacions d'aquestes plantes, però depèn de la presència d'ocells frugívors. L'eliminació d'elements singulars de l'hàbitat o *legacies* com ara els arbres aïllats que poden atraure ocells dispersors de llavors (Guevara & Laborde 1993, Pausas *et al.* 2006, Herrera & García 2009) podria tenir un efecte negatiu per al manteniment de la dispersió per ocells frugívors. En aquest sentit, Castro *et al.* (2010) van observar que en pinedes cremades de muntanya (a Sierra Nevada, Andalusia), la presència de túrdids, els principals responsables de la dispersió de llavors de fruits carnosos en aquella àrea, es veia afectada molt negativament per la tala post-incendi. En conseqüència, la dispersió de llavors que duen a terme aquests ocells també es podria veure compromesa.

Per tant, el conjunt d'estudis existents posa de manifest que la tala post-incendi i altres formes concretes de gestió de les restes cremades afecten de manera important la presència i distribució de moltes espècies d'animals. A més a més, aquests tipus de gestió també influeix en l'ús que fa la fauna dels recursos, la qual cosa pot tenir, en determinats casos, repercussions importants en altres processos a nivell d'ecosistema on intervenen animals, com la regeneració de la vegetació.

Objectius

Els ocells són un grup faunístic que es considera un bon indicador dels canvis en l'hàbitat i en el paisatge, per diverses raons: moltes espècies responen a aquests canvis de forma ràpida, els seus requeriments d'hàbitat són força predictibles en època de cria, hi ha molta informació sobre la seva biologia, distribució i tendències, etc. (Furness & Greenwood 1993, Bibby 1999, Robledano *et al.* 2010). A més a més, certes espècies estan implicades en processos clau per al funcionament dels ecosistemes mediterranis, com la dispersió de llavors de plantes amb fruits carnosos (Herrera 1984). Per això, aquest grup és un bon model d'estudi per avaluar l'impacte dels canvis que té la tala post-incendi a curt terme sobre la fauna i sobre el procés de dispersió de llavors fet per la fauna frugívora en àrees cremades.

La tala post-incendi suposa un impacte important en l'avifauna de boscos cremats, que molts estudis previs han valorat com a negatiu per a la conservació. Tanmateix, la majoria d'aquests estudis s'han dut a terme fora d'Europa, on els ocells s'enfronten a problemàtiques força diferents a les d'altres zones. Per exemple, moltes espècies lligades a ambients oberts i agrícoles es troben en declivi pels canvis en els usos del sòl que han tingut lloc a Europa (inclosa la part mediterrània) en les últimes dècades. La tala post-incendi, per tant, podria beneficiar també algunes d'aquestes espècies, pel canvi en l'hàbitat que porta associat (de forestal a obert). L'objectiu del **capítol 1** és estudiar quins elements de l'hàbitat cremat tenen més efecte sobre la composició de l'avifauna, tant a nivell de comunitat com d'espècie, i de manera particular sobre els ocells amb interès de conservació. Per això, s'analitzen quins components de l'hàbitat poden tenir un efecte més important, si aquells relacionats amb la tala (densitat d'arbres) o aquells relacionats amb la regeneració de la vegetació (recobriment vegetal, presència de rieres). A més, també s'analitza l'efecte del pas del temps i de la gestió sobre la comunitat d'ocells mitjançant una aproximació pseudo-experimental BACI (before/after control-impact).

Tot i ser una pràctica estesa en boscos de tot el món, l'efecte de la tala post-incendi sobre comunitats i espècies d'ocells ha estat estudiat sobretot a les muntanyes Rocalloses de Nord Amèrica. Manquen, en canvi, estudis que comparin aquest efectes

Figura 5. Bosc de coníferes cremat en el foc de Black Mountain (Montana) l'agost de 2003. La fotografia està presa gairebé set anys després, el juny de 2010.

entre diferents parts del món i dels que se'n puguin derivar generalitzacions. La Conca Mediterrània, per exemple, és una regió on els incendis forestals són una pertorbació recurrent, amb un paisatge i unes comunitats d'ocells amb unes característiques ben diferents a les de les Rocalloses. L'objectiu del **capítol 2** és explorar els patrons de resposta a la tala post-incendi per part de grups d'ocells amb requeriments similars pel que fa a hàbitat, nidificació i alimentació, entre les muntanyes Rocalloses i la Conca Mediterrània, per tal de trobar patrons similars que permetin generalitzar els efectes d'aquesta gestió a una escala intercontinental. A més, també es pretén valorar si els efectes de la tala post-incendi es poden considerar diferents a l'hora de donar recomanacions sobre la gestió. Per fer això es van utilitzar bases de dades extenses sobre censos d'ocells en boscos de coníferes cremats de Catalunya (regió mediterrània) i de Montana (muntanyes Rocalloses), basades en mostrejos realitzats durant el tercer i quart any després del foc.

D'altra banda, els canvis en l'hàbitat fruit de la gestió post-incendi també poden afectar la composició de la comunitat d'ocells a hivern, que és l'època principal en què té lloc el procés de dispersió de llavors de plantes amb fruit carnós per mitjà d'ocells frugívors. Si el canvi en l'hàbitat afectés la presència de determinades espècies d'ocells frugívors implicades en la dispersió de llavors de plantes amb fruits carnosos, de retruc, podria afectar –negativament o positiva- al procés de dispersió. L'objectiu del **capítol 3** és estudiar l'efecte de la tala post-incendi sobre la comunitat d'ocells a l'hivern i analitzar quines parts de l'hàbitat resultant d'aquesta gestió (arbres cremats morts no tallats, arbres vius no tallats, vegetació arbustiva, sòl nu i pilons de branques) són més utilitzades pels ocells frugívors implicats en el procés de dispersió de plantes amb fruit carnós, i si aquests ocells seleccionen positivament algun d'aquests microhàbitats, la qual cosa pot derivar en propostes concretes de gestió.

La dispersió de llavors per ocells des de zones no cremades fins a l'interior de boscos cremats és un procés que podria ser d'importància per a la regeneració de la vegetació de les àrees cremades, i que tanmateix no ha estat descrit. D'altra banda, l'ús preferencial que poden fer els ocells frugívors de les noves estructures o microhàbitats originats per la gestió post-incendi, hipotèticament podria generar una pluja de llavors més intensa sota algun d'ells, en particular sota d'arbres, que podrien atreure espècies frugívores. La intensitat de la pluja de llavors, a més a més, pot dependre de la distància a la qual es trobi la font de fruits, en zones no cremades. Aquestes preguntes s'intenten respondre al **capítol 4**, mesurant la pluja de llavors de plantes amb fruit carnós dins del perímetre d'un incendi per mitjà de trampes de recollida de llavors situades sota arbres aïllats, arbres cremats situats en zones no talades, sòl nu, vegetació arbustiva i pilons de branques. Per assegurar que les llavors provenien de zones no cremades, el mostreig es va realitzar el segon hivern després del foc, aprofitant que encara no hi havia producció de fruits dins de la zona cremada i que els microhàbitats resultants de la tala post-incendi ja estaven construïts.

Finalment, havent vist que efectivament la intensitat de la dispersió de llavors de plantes amb fruit carnós depèn de les estructures originades per la gestió forestal post-incendi (a
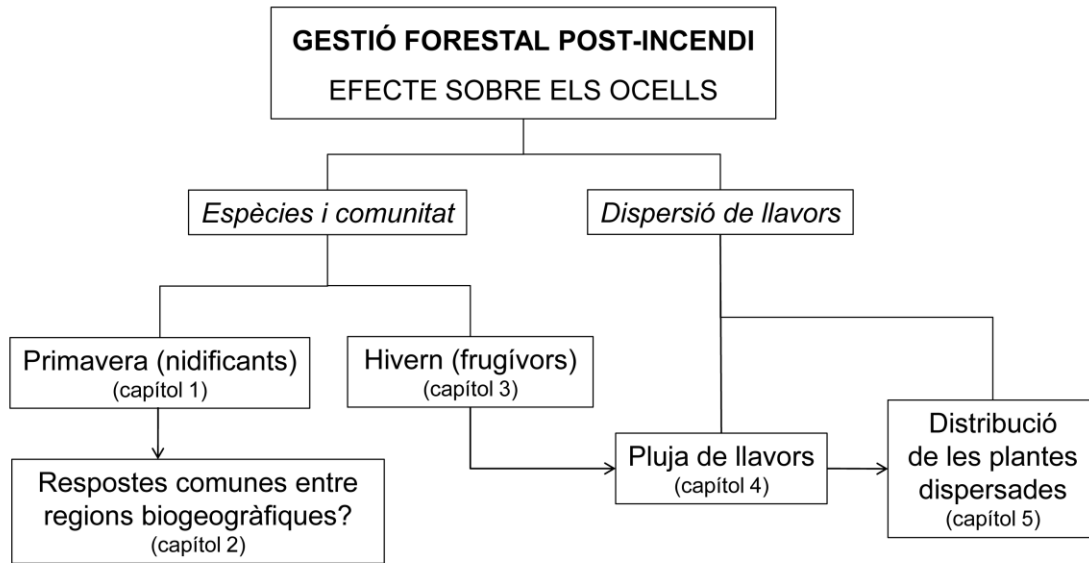
Figura 6. Resum dels objectius de la tesi.

partir dels resultats dels capítols 3 i 4), essent més intensa en pilons de branques que en altres zones, queda per respondre la qüestió de la qualitat d'aquesta dispersió. És a dir, si la diferent pluja de llavors en aquests pilons dóna com a resultat una major abundància de les plantes dispersades per ocells frugívors en aquests pilons. Verificar aquesta hipòtesi és l'objectiu del **capítol 5**, en el qual es va analitzar si el recobriment de plantes emergides, productores de fruits carnosos, és més elevat en pilons que fora de pilons. Aquest estudi es va realitzar en quatre incendis diferents, tots ells de pi blanc.

Àrea d'estudi

L'àrea d'estudi d'aquesta tesi es troba a la terra baixa de Catalunya; un territori que es caracteritza per un paisatge format per un mosaic de zones agrícoles i boscos dominats pel pi blanc. Aquest és un tipus de bosc que colonitza fàcilment zones pertorbades o camps de conreu abandonats (Blondel & Aronson 1999), la qual cosa ha fet augmentar la seva superfície de manera significativa en les darrers cent anys, convertint-se en l'espècie arbòria més abundant del territori català (Burriel *et al.* 2004). D'altra banda, aquests tipus de boscos són els més afectats per incendis forestals a Catalunya (Zozaya *et al.* 2011), després dels quals el pi blanc regenera vigorosament, de manera que en una vintena d'anys ja pot haver-hi una nova pineda sense cap necessitat de repoblacions. Després del foc, la gestió més habitual que té lloc als boscos de pi blanc és la tala dels arbres. En alguns incendis, especialment en aquells de boscos públics, les restes no comercials s'han apilonat per a construir feixines, amb l'objectiu de reduir l'erosió.

En concret, el treball de camp per als treballs que es presenten s'ha fet a la comarca de l'Alt Empordà, als incendis de Ventalló (613 ha forestals cremades) i Cistella (170 ha), de l'agost de 2006 (capítols 1 i 3), i al de Santa Caterina, al Montgrí (188 ha), del

setembre de 2004 (capítols 3, 4 i 5). Altres incendis mostrejats durant la tesi són el de Cardona (1.438 ha), al Bages, i el de Sant Llorenç Savall (4.559 ha), al Vallès Occidental (capítol 5)

D'altra banda, l'àrea d'estudi per al segon capítol inclou 23 incendis diferents a tot Catalunya, mostrejats per diferents observadors en el marc del projecte DINDIS (incendis ocorreguts entre els anys 2000 i 2007, inclosos els esmentats abans; per detalls veure Zozaya *et al.* 2010) i 17 incendis a Montana, als Estats Units (tots ells de l'any 2003), també mostrejats per diversos observadors.

Figura 7. Situació a Catalunya dels incendis on es va dur a terme més treball de camp durant la tesi.

Referències

Amman, G.D. & Ryan, K.C. 1994. Insect infestation of fire-injured trees in the Greater Yellowstone area. Research Note INT-398. USDA Forest Service, Missoula.

Arnan, X., Rodrigo, A. & Retana, J. 2006. Post-fire recovery of Mediterranean ground ant communities follows vegetation and dryness gradients. Journal of Biogeography 33, 1246-1258.

Baker, W.L. 2009. Fire ecology in Rocky Mountain landscapes. Island Press, Washington DC.

Bautista, S., Gimeno, T., Mayor, G.A. & Gallego, D. 2004. El tratamiento de la madera quemada tras los incendios forestales. In: Vallejo, R.A. & Alloza, J.A. (eds.). Avances en el estudio de la gestión del monte mediterráneo. Fundación Centro de Estudios Ambientales del Mediterráneo (CEAM), Valencia. Pp. 547-570.

Barlow, J. & Peres, C.A. 2004. Avifaunal responses to single and recurrent wildfires in Amazonian forests. Ecological Applications 14, 1358-1373.

Beschta, R.L., Rhodes, J.J., Kauffman, J.B., Gresswell, R.E., Minshall, G.W., Karr, J.R., Perry, D.A. Hauer, F.R. & Frissell, C.A. 2004. Postfire management on forested public lands of the western United States. Conservation Biology 18, 957-967.

Bibby, C.J., 1999. Making the most of birds as environmental indicators. Ostrich 70, 81-88.

BirdLife International. 2004. Birds in Europe: population estimates, trends and conservation status. BirdLife International, Wageningen.

Blondel, J. & Aronson, J. 1999. Biology and wildlife of the Mediterranean region. Oxford University Press, New York.

Brotons, L. Pons, P. & Herrando, S. 2005. Colonization of dynamic Mediterranean landscapes: where do birds come from after fire? Journal of Biogeography 32, 789-798.

Burriel, J.A., Gracia, C., Ibàñez, J.J., Mata, T., & Vayreda, J. 2004. Inventari Ecològic i Forestal de Catalunya. Catalunya. CREAF, Bellaterra.

Cahall, R.E. & Hayes, J.P. 2009. Influences of postfire salvage logging on forest birds in the Eastern Cascades, Oregon, USA. Forest Ecology and Management 257, 1119-1128.

Castro, J., Navarro, R., Guzmán, J.R., Zamora, R. & Bautista, S. 2009. ¿Es conveniente retirar la madera quemada tras un incendio forestal? Quercus 281, 34-41.

Castro, J., Moreno-Rueda, G. & Hódar, J.H. 2010. Experimental test of post-fire management in pine forests: impact of salvage logging versus partial cutting and non-intervention on bird-species assemblages. Conservation Biology 24, 810-819.

Cunningham, S.C., Kirkendall, L. & Ballard, W. 2006. Gray fox and coyote abundance and diet responses after a wildfire in central Arizona. Western North American Naturalist 66, 169-180.

Dellasala, D.A., Karr, J.R., Schoennagel, T., Perry, D., Noss, R.F., Lindenmayer, D., Beschta, R., Hutto, R.L., Swanson, M.E. & Evans, J. 2006. Post-fire logging debate ignores many issues. Science 314, 51-52.

Eckberg, T.B., Schmid, J.M., Mata, S.A. & Lundquist, J.E. 1994. Primary focus for the mountain pine beetle in the Black Hills. Research Note RM-RN-531. USDA Forest Service, Fort Collins.

Fattorini, S. 2010. Effects of fire on tenebrionid communities of a Pinus pinea plantation: a case study in a Mediterranean site. Biodiversity and Conservation 19, 1237-1250.

Fisher, J.T. & Wilkinson, L. 2005. The response of mammals to forest fire and timber harvest in the North American boreal forest. Mammal Review 35, 51-81.

Folch, R. & Castelló, J.I. 1976. Cuadernos de ecología aplicada: los incendios forestales. Diputación de Barcelona, Barcelona.

Furness, R.W. & Greenwood, J.J.D. 1993. Birds as monitors of environmental change. Chapman & Hall, London.

Guevara, S., Laborde, J., 1993. Monitoring seed dispersal at isolated standing trees in tropical pastures – consequences for local species availability. Vegetatio 108, 319-338.

Haim, A. & Izhaki, I. 1994. Changes in rodent community during recovery from fire: relevance to conservation. Biodiversity and Conservation 3, 573-585.

Henriques, R.P.B., Briani, D.C., Palma, A.R.T. & Vieira, E.M. 2006. A simple graphical model of small mammal succession after fire in the Brazilian cerrado. Mammalia, 226-230.

Herrando, S., Brotons, L. & Llacuna, S. 2003. Does fire increase the spatial heterogeneity of bird communities in Mediterranean landscapes? Ibis 145, 307-317.

Herrando, S., Brotons, L., Guallar, S., Sales, S. & Pons, P. 2009. Postfire forest management and Mediterranean birds: the importance of logging remnants. Biodiversity and Conservation 18, 2153-2164.

Herrera, C.M. 1984. A study of avian frugivores, bird-dispersed plants, and their interaction in Mediterranean scrublands. Ecological Monographs 54, 1-23.

Herrera, J.M. & García, D. 2009. The role of remnant trees in seed dispersal through the matrix: being alone is not always so sad. Biological Conservation 142, 149-158.

Hessburg, P.F. & Agee, J.K. 2003. An environmental narrative of Inland Northwest United States forests, 1800-2000. Forest Ecology and Management 178, 23-59.

Hoyt, J.S., Hannon, S.J. 2002. Habitat associations of black-backed and three-toed woodpeckers in the boreal forest of Alberta. Canadian Journal of Forest Research 32, 1881-1888.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9, 1041-1058.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation Biology 20, 984-993.

Hutto, R.L. & Gallo, S.M. 2006. The effects of postfire salvage logging on cavity-nesting birds. The Condor 108, 817-831.

Hutto, R.L. 2008. The ecological importance of severe wildfires: some like it hot. Ecological Applications 18, 1827-1834.

Imbeau, L., Savard, J-P.L. & Gagnon, R. 1999. Comparing bird assemblages in successional spruce stands originating from fire and logging. Canadian Journal of Zoology 77, 1850-1860.

Izhaki, I. & Adar, M. 1997. The effects of postfire management on bird community succession. International Journal of Wildland Fire 7, 335-342.

Jacquet, K. & Prodon, R. 2009. Measuring the postfire resilience of a bird-vegetation system: a 28-year study in a Mediterranean oak woodland. Oecologia 161, 801-811.

Leidoff, A. & Bissonette, J.A. 2009. The effects of fire on avian communities: spatio-temporal attributes of the literature 1912-2003. International Journal of Wildland Fire 18, 609-622.

Ley 10/2006, de 28 de abril, por la que se modifica la Ley 43/2003, de 21 de noviembre, de Montes. BOE núm. 102 del 29/4/2006. http://www.boe.es/aeboe/consultas/bases_datos/-doc.php?id=BOE-A-2006-7678.

Lindenmayer, D.B., Foster, D.R., Franklin, J.F., Hunter, M.L., Noss, R.F., Schimegelow, F.A. & Perry, D. 2004. Salvage harvesting policies after natural disturbance. Science 303, 1303.

Lindenmayer, D.B., Wood, J.T., Cunningham, R.B., MacGregor, C., Crane, M., Michael, D., Montague-Drake, R., Brown, D., Muntz, R. & Gill, A.M. 2008. Testing hypotheses associated with bird responses to wildfire. Ecological Applications 18, 1967-1983.

Lindenmayer, D.B., Wood, J.T., MacGregor, C., Michael, D.R., Cunningham, R.B., Crane, M., Montague-Drake, R., Brown, D., Muntz, R. & Driscoll, D.A. 2008. How predictable are reptile responses to wildfire? Oikos 117, 1086-1097.

Llimona, F., Matheu, E., Prodon, R. 1993. Role of snag persistence and of tree regeneration in postfire bird successions: comparison of pine and oak forests in Montserrat (Catalonia, NE Spain). In: Trabaud, L., Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Bruxelles. Pp. 315-331.

Loeb, S.C. 1999. Responses of small mammals to coarse woody debris in a southeastern pine forest. Journal of Mammalogy 80, 460-471.

Kaynaş, B.Y. & Gürkan, B. 2005. Changes in Buprestidae (Coleoptera) community with successional age after fire in a *Pinus brutia* forest. Journal of Pest Science 78, 53-55.

Keane, R.E., Ryan, K.C., Veblen, T.T., Allen, C.D., Logan, J. & Hawkes, B. 2002. Cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. General Technical Report RMRS-GTR-91. USDA Forest Service, Fort Collins.

Kotliar, N.B., Hejl, S.J., Hutto, R.L., Saab, V., Melcher, C.P. & McFadzen, M.E. 2002. Effects of fire and postfire salvage logging on avian communities in conifer dominated forests of the western United States. Studies in Avian Biology 25, 49-64.

Kulakowski, D. & Veblen, T.T. 2007. Effect of prior disturbances on the extent and severity of wildfire in Colorado subalpine forests. Ecology 88, 759-769.

Krawchuk, M.A., Moritz, M.A., Parisien, M-A., Jeff Van Dorn, J. & Hayhoe, K. 2009. Global Pyrogeography: the Current and Future Distribution of Wildfire. Plos One 4, e5102. doi:5110.1371/journal.pone.0005102

Manning, J.E. & Edge, W.D. 2005. Small mammal responses to fine woody debris and forest fuel reduction in southwest Oregon. The Journal of Wildlife Management 72, 625-632.

Martikainen, P., Kouki, J., Heikkala, O., Hyvärinen, E. & Lappalainen, H. 2006. Effects of green tree retention and prescribed burning on the crown damage caused by the pine shoot beetles (*Tomicus* sp) in pine-dominated timber harvest areas. Journal of Applied Entomology 130, 37-44.

McCullogh, D.G., Werner, R.A. & Neumann, D. 1998. Fire and insects in northern and boreal forest ecosystems of North America. Annual Review of Entomology 46, 107-127.

McIver, J.D. & Starr, L. 2000. Environmental effects of postfire logging: literature review and annotated bibliography. General Technical Report PNW-GTR-486. USDA Forest Service, Portland.

Moneglia, P., Besnard, A., Thibault, J-C. & Prodon, R. 2009. Habitat selection of the Corsican Nuthatch (*Sitta whiteheadi*) after a fire. Journal of Ornithology 150, 577-583.

Moreira, F., Ferreira, P.G., Rego, F.C. & Bunting, S. 2001. Lanscape changes and breeding bird assemblages in northwestern Portugal: the role of fire. Landscape Ecology 16, 175-187.

Morisette, J.L., Cobb, T.P., Brigham, R.M. & James, P.C., 2002. The response of boreal songbird communities to fire and postfire harvesting. Candian Journal of Forest Research 32, 2169-2183.

Pausas, J.G., Bonet, A., Maestre, F.T., Climent, A. 2006. The role of the perch effect on the nucleation process in Mediterranean semi-arid oldfields. Acta Oecologica 29, 346-352.

Pausas, J.G., Llovet, J., Rodrigo, A. & Vallejo, R. 2008. Are wildfires a disaster in the Mediterranean basin? –A review. International Journal of Wildland Fire 17, 713-723.

Pausas, J.G. & Keeley, J.E. 2009. A burning story: the role of fire in the history of life. BioScience 59, 593-601.

Piñol, J., Terradas, J., & Lloret, F. 1998. Climate warming, wildfire hazard, and wildfire occurrence in coastal Eastern Spain. Climatic Change 38, 345-357.

Plana, E. 2004. Anàlisi d'escenaris de prevenció i extinció d'incendis des de la perspectiva socioambiental. A: Incendis forestals, dimensió socioambiental, gestió del risc i ecologia del foc. Plana, E. (ed.). Xarxa ALINFO, Solsona. Pp. 5-12.

Pons, P. & Prodon, R. 1996. Short term temporal patterns in a Mediterranean shrubland bird community after wildfire. Acta Oecologica 17, 29-41.

Pons, P. & Bas, J.M. 2005. Open-habitat birds in recently burned areas: the role of the fire extent and species' habitat breadth. Ardeola 52, 119-131.

Prodon, R. & Pons, P. 1993. Postfire bird studies: methods, questions and perspectives. In: Trabaud, L. & Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Commission of the European Communities, Brussel·les. Pp. 332-343.

Resolució MAH/965/2006, de 5 d'abril. DOGC núm. 4612, de l'11 d'abril de 2006. http://www.gencat.cat/diari/4612/06093142.htm

Robledano, F., Esteve, M.A., Farinós, P., Carreño, M.F., & Martínez-Fernández, J. 2010. Terrestrial birds as indicators of agricultural-induced changes and associated loss in conservation value of Mediterranean wetlands. Ecological Indicators 10, 274-286.

Rodrigo, A. & Retana, J. 2006. Post-fire recovery of ant communities in Submediterranean *Pinus nigra* forests. Ecography 29, 231-239.

Rodrigo, A., Sardà-Palomera, F., Bosch, J. & Retana, J. 2008. Changes of dominant ground beetles in black pine forests with fire severity and successional age. Écoscience 15, 442-452.

Rollán, A. & Real, J. 2010. Effect of wildfires and post-fire forest treatments on rabbit abundance. European Journal of Wildlife Research, in press. DOI: 10.1007/s10344-010-0412-y.

Saab, V. A., & J. G. Dudley. 1998. Responses of cavity nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. Research Paper RMRS-RP-11. USDA Forest Service, Ogden.

Saab, V., Russell, R.E. & Dudley, J.G. 2007. Nest densities of cavity-nesting birds in relation to postfire salvage logging and time since wildfire. The Condor 109, 97-108.

Santos, X., Bros, V. & Miño, A. 2009. Recolonization of a burned Mediterranean area by terrestrial gastropods. Biodiversity and Conservation 18, 3153-3165.

Santos, X. & Poquet, J.M. 2010. Ecological succession and hábitat attributes affect the postfire response of a Mediterranean reptile community. European Journal of Wildlife Research 56, 895-905.

Schmiegelow, F.K.A., Stepnisky, D.P., Stambaugh, C.A. & Koivula, M. 2006. Reconciling salvage logging of boreal forests with a natural-disturbance management model. Conservation Biology 20, 971-983.

Smucker, K.M., Hutto, R.L. & Steele, B.M. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15, 1535-1549.

Thomson, J.R., Spies, T.A. & Ganio, L.M. 2007. Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Sciences 19, 10743-10748.

Tiedemann, A.R., Klemmedson, J.O. & Bull, E.L. 2000. Solution of forest health prolems with prescribed fire: are forest productivity and wildlife at risk? Forest Ecology and Management 127, 1-18.

Vallecillo, S., Brotons, L. & Herrando, S. 2008. Assessing the response of open-habitat bird species to landscape changes in Mediterranean mosaics. Biodiversity and Conservation 17, 103-119.

Zozaya, E.L., Brotons, L., Herrano, S., Pons, P., Rost, J. & Clavero, M. 2010. Monitoring bird community dynamics in Mediterranean landscapes affected by large wildfires. Ardeola 57, 33-50.

The effect of postfire salvage logging on bird communities in Mediterranean pine forests: the benefits for declining species

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The effect of postfire salvage logging on bird communities in Mediterranean pine forests: the benefits for declining species

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Summary

1. Postfire salvage logging is the forestry practice most commonly applied in burned forests worldwide, mainly for economic reasons. However, it strongly affects bird communities and is generally considered to be detrimental for bird conservation. In the Mediterranean part of Europe, as elsewhere in the continent, birds face different threats to those encountered in less intensely exploited areas (e.g., the loss of habitat for declining open-habitat species due to recent land-use changes). We therefore aimed to assess the effect of postfire salvage logging on birds in the Mediterranean basin, with particular reference to declining species.

2. We surveyed breeding birds in two burned secondary pine forests in Catalonia (western Mediterranean Basin), and analyzed the effect of salvage logging (measured through snag density) and vegetation regeneration as determinants of individual species and community parameters. We also used a pseudo-experimental before-after-control-impact approach to study the changes in the bird community during the first three springs after fire.

3. Most bird species were affected by snag density, which was positive for forest birds and negative for open-habitat species, although those more linked to shrub and edge habitats were more affected by vegetation regrowth. Logged areas showed a higher conservation value than unlogged areas. Species richness and overall density were affected significantly by time since fire.

4. Salvage logging is the main factor affecting the occurrence of bird species in burned Mediterranean pine forests. It benefits a number of declining open-habitat species, although it is not positive for all species of conservation concern. In contrast to published literature, we show that the effect of salvage logging on bird conservation cannot always be considered as negative; it depends on the specific threats that declining birds face in each region.

5. *Synthesis and applications*. The results of this study suggest that a certain level of postfire wood harvesting can be compatible with bird conservation in the Mediterranean region. Such management should be planned taking into consideration the preservation of the ecosystem's functionality. We make recommendations along these lines that may be of help to forest managers and policy makers.

Keywords: colonization, conservation, disturbance, fire, open-habitat birds, forest management, wood harvesting.

Introduction

Salvage logging has traditionally been the most common management practice carried out in forests affected by severe wildfires worldwide (Lindenmayer *et al*. 2004; Hutto 2006; Lindenmayer & Noss 2006). The main goal of postfire salvage logging is to minimize or recoup the economic loss caused by fire on timber, by selling surface-charred logs before they start to deteriorate and become unsalable. However, other justifications have been given for removing dead wood, such as speeding forest recovery, avoiding the proliferation of xylophagous insects, reducing fuel for future fires, making transit within the burned area easier and safer, reducing erosion with log barriers, and even improving the aesthetic appearance of the forest (Ne'eman *et al.* 1995; Martínez-Sánchez *et al.* 1999; Sessions 2004). There has been very little social resistance to these justifications because the prevailing perception is that wildfires (especially severe wildfires) are catastrophic events that are not part of the natural dynamics of ecosystems and that burned areas have little value to wildlife (Ne'eman *et al.* 1997; Hessburg & Agee 2003; Lindenmayer & Noss, 2006; Pausas *et al.* 2008). This view, together with the associated belief that burned areas need to be rehabilitated, promotes the generally accepted idea that salvage logging is a positive practice that may even accelerate forests' return to pre-fire 'ideal' conditions.

However, postfire logging has a strong impact on ecosystem processes involving soil, nutrient and hydrological cycles, and also on the ecosystem's physical structure, since the removal of snags creates a drastic habitat change from a burned forest structure to an unwooded area (McIver & Starr, 2000; Beschta *et al.* 2004; Karr *et al.* 2004; Dellasala *et al.* 2006). This habitat transformation has strong effects on the biota living in burned areas. Studies carried out on birds show a reduction of forest species living in logged areas compared with those in unlogged ones, because of the removal of snags that provided them food and nesting opportunities (Llimona *et al.* 1993; Morisette *et al.* 2002; Hutto 2006; Stuart-Smith *et al.* 2006; Cahall & Hayes 2009; Castro *et al.* 2009; Moneglia *et al.* 2009). In western North American forests, salvage logging is especially detrimental for certain woodpeckers that are considered fire specialists due to their strong dependence on severely burned stands (Hutto & Gallo 2006; Koivula & Schmiegelow 2007; Hutto 2008). At the same time, partial or moderate salvage logging may create habitats for species that live in more open environments (Kotliar *et al.* 2002;

Comparing the effect of salvage logging on birds in the Mediterranean basin and the Rocky Mountains: common patterns, different management implications

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Conservation Biology (en revisió)

Comparing the effect of salvage logging on birds in the Mediterranean basin and the Rocky Mountains: common patterns, different management implications

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Abstract

Postfire salvage logging is currently a controversial issue because of the impact it has been argued to have on forest ecosystems. Although it is a relatively extended practice worldwide, its effects have been scarcely studied and the absence of comparisons across regions hinders the development of broad generalizations. Here we compared bird response to postfire salvage logging in two regions with important differences in landscape and bird communities, the Mediterranean basin and the Rocky Mountains, aiming to find general responses of species with similar life-history traits when facing salvage logging. We used databases describing bird species occurrence in burned forests from both regions and defined three groups of birds according to their increasing dependence on snags. We then analyzed the differences in the proportion of each group between samples that were heavily logged, lightly logged or unlogged, and also between regions. Species that showed some degree of dependence on snags were more common in burned forests of the Rocky Mountains than in the Mediterranean basin, and the opposite held true for non-snag-dependent species. Highly snag-dependent birds from both regions showed a common negative response to salvage logging. Non-snag-dependent species increased in salvaged areas, but only in the Mediterranean basin. The general negative effect of salvage logging on forest-dwelling species that depend on trees or snags is a noteworthy pattern given the big differences between regions. However, considering that forest birds, including several fire specialists, are especially important in the Rocky Mountains and that in the Mediterranean some threatened open-habitat species even benefit from logging, we considered the overall effect of the removal of snags to be more detrimental in the Rocky Mountains. Therefore, we recommend concentrating wood harvesting efforts in less sensitive areas rather than in burned forests in the Rocky Mountain region.

Keywords: Catalonia, conifer forests, conservation, fire, forest management, Montana, wood harvesting.

Introduction

Wildfires are a crucial part of the natural dynamics of many forest ecosystems worldwide. Species living in environments that have been repeatedly affected by fire show adaptations to a particular disturbance regime; in some instances, species even appear to have evolved to depend on the presence of relatively intense fire events (Hutto 2008, Pausas & Keeley 2009). Although wildfire is a natural component of the forest dynamics in most systems, in areas where forests are managed for wood harvesting or other purposes, wildfire is usually considered to be a problem mainly because the loss of economic value associated with timber. Consequently, the prevailing public perception in many countries is still that wildfires are environmental disasters that have to be avoided, and that burned forests must be "restored" or rehabilitated to re-create "healthy" forests as rapidly as possible (Swanson *et al.* 2010). This public perception has also led to a level of fire suppression that has affected forest structure and dynamics (Hessburg & Agee 2003, Pausas *et al.* 2008), and to current postfire management practices that emphasize salvage logging over a hands-off management strategy during the most sensitive phase of forest succession. In private forests aimed at timber production, postfire logging allows the landowner to recoup some of the economic losses caused by fire, which seems reasonable, given the primary objective of the landowner. In public forests, however, ecological sustainability is assumed to be the prevailing goal and therefore salvage logging for the sake of mining the economic value tied up in the standing dead timber is not as easily justified. After all, there are values associated with a standing dead forest to consider. Specifically, snags and other coarse woody debris play important roles in the ecosystem after fire, being sources of nutrients, providing habitat to species that need burned forest conditions, and contributing to the role of natural succession in the ecosystem (McIver & Starr 2000, Lindenmayer & Noss 2006). Nevertheless, public land managers tend to couple economic values with a perceived need for "forest restoration" to gain public support for salvage logging operations.

The removal of snags during salvage logging operations profoundly affects the process of natural forest succession, however, and can be viewed as a second form of disturbance rather than a restoration activity. How do organisms normally respond to natural disturbance events, and how does the occurrence of a second form of artificial disturbance affect the response of those organisms to the natural disturbance event? The impact of salvage logging on the biota living in burned forests has been studied mostly with birds, which show responses that differ among species (Llimona *et al.* 1993, Kotliar *et al.* 2002, Morisette *et al.* 2002). In general, forest birds are negatively affected by the removal of snags, while early-successional species may benefit from this practice. Interestingly, most studies that have focused on the issue of salvage logging have been carried out primarily in conifer forests from two regions: northwestern United States and Canada (e.g. see Saab & Dudley 1998, Kotliar *et al.* 2002, Morisette *et al.* 2002, Saab *et al.* 2007, Hutto 2008, Cahall & Hayes 2009) and, to a lesser extent, the Mediterranean basin (e.g. Llimona *et al.* 1993; Izhaki & Adar 1997; Rost *et al.* 2010;

Castro *et al.* 2010). For millennia, wildfires have played (and still play) an important role in the vegetation and landscape dynamics of both regions (Arno *et al.* 2000, Hessburg & Agee 2003, Pausas *et al.* 2008), which also share salvage logging as the most commonly performed management treatment after a conifer forest burns.

However, these two regions present striking differences concerning other aspects of forest management, fire regime, human settlement and land-use histories, and the landscape configuration and composition of their bird communities. North American forests have undergone lower human pressure and management intensity than those in the Mediterranean basin (Hessburg & Agee 2003), and they harbor a higher number of forest specialist birds, including several fire specialists (Hutto 2006). In contrast, Mediterranean forests have been deeply modified by humans for ages (Pausas 2008), and their forest bird communities are basically composed of species that are widely distributed across most European forests (Blondel & Aronson 1999). On the other hand, the bird species that have captured most conservationists' concerns and efforts in Europe in recent decades are those from steppes, shrublands and farmlands, which have sharply declined due to habitat loss and degradation (BirdLife International 2004). Some of these species colonize recently burned areas, benefitting from fire (Pons & Bas 2005, Brotons *et al.* 2008) and logging (Castro *et al.* 2010, Rost *et al.* 2010). The differences in both regions' avifauna need to be considered when assessing the impact of salvage logging on their bird communities.

In this study, we compare the response of birds from these two distant regions to postfire salvage logging, aiming to find common patterns that would help us to understand and generalize the effects of postfire forest management in a way that could overcome geographical and political boundaries. We hypothesize that birds with similar life-history traits concerning their use and dependence of trees and snags will probably share common responses to the removal of snags in a burned forest. On the other hand, given the differences between the avifaunas from both regions and their threats, we also predicted that the impact of postfire salvage logging would be more severe in the Rocky Mountains than in the Mediterranean basin. With these questions in mind, we took advantage of the availability of information on the occurrence of birds in burned areas gathered in Catalonia (Mediterranean basin) and western Montana (Rocky Mountains), and studied the response of groups of bird species from both regions to salvage logging after fire.

Methods

Study area

Catalonia, located in the western part of the Mediterranean basin (henceforth WMB) is a 32,000 km^2 region located in the northeastern corner of the Iberian Peninsula (latitudinal range from 40ºN to 42ºN; Figure 1). Excluding the temperate and colder
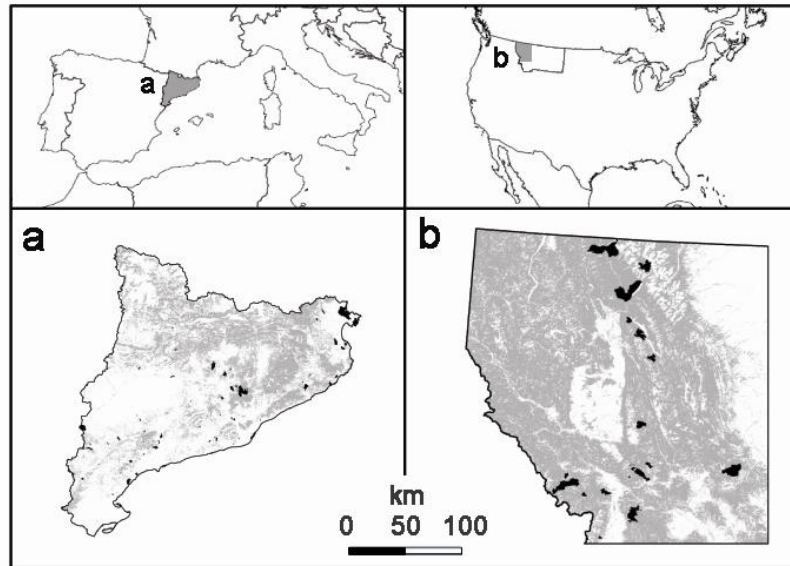
Figure 1. Location of the two regions compared: a) West Mediterranean Basin (Catalonia) and b) Northern Rocky Mountains (north-western Montana). In the enlarged pictures, which are in the same scale, studied burned areas that were used in the comparison are highlighted in black. Areas darkened in gray represent forest cover, which are shown in order to highlight landscape configuration differences between the two regions.

Pyrenees Mountains in the north and northwest, which are forested but where wildfires are very rare, the rest of Catalonia has a typically Mediterranean climate and a rugged landscape dominated by a mosaic of farmland, forest and shrubland. These low-elevation forests are dominated by Aleppo pines (*Pinus halepensis*), European black pines (*Pinus nigra*) and Holm oaks (*Quercus ilex*). Most of them are second-growth forests that have colonized former agricultural areas.

The northern Rocky Mountain study area (henceforth NRM) is a 80,000 km^2 region located in the northwestern part of the United States, in western Montana (latitude between 46ºN and 48ºN). The region's climate is continental, and the landscape is dominated by mountains covered by conifer forests, whose main species are Ponderosa pine (*Pinus ponderosa*), Lodgepole pine (*Pinus contorta*) and Douglas-fir (*Pseudotsuga menziesii*). Sagebrush, prairies and farmland are found mostly in valleys. In contrast with the Catalan system, the Rocky Mountain system is largely intact, and many forests grow as they have for millennia.

The plant species of both regions show several common adaptations to fire, which reflect the fact that they may have evolved with a long history of fire (Pausas and Keeley 2009). There are fire-dependent pines with serotinous cones that open and release the seeds in the presence of fire (Aleppo pine in WMB, Lodgepole pine in NRM), there are species with thick bark, high canopy and few low branches that avoid torching, that can survive surface fires (European black pine in WMB, Ponderosa pine in NRM), and even some species that resprout after being burned if fire does not kill

them outright (Holm oak and Cork oak (*Quercus suber*) in WMB, Western larch (*Larix occidentalis*) in NRM).

However, there are fundamental differences in the fire regimes between the two regions. In WMB, most fires are human-caused, while in NRM most of them ignite by natural causes (lightning). This is not surprising, given that the population density in Catalonia (about 230 people/km^2) is roughly 50 times that of western Montana. The average size of fires is much smaller in WMB than in NRM (Table 1), which is partly a consequence of the differences in landscape configuration between the two regions. Most fires in WMB are high-severity, stand-replacement fires, which have increased in number in recent decades (Pausas 2004, González & Pukkala 2007) due to rural depopulation, farmland activity abandonment (Pausas *et al.* 2008), and fire exclusion policies that contributed to increases in fuel loads and forest connectivity. In NRM, most fires are mixed-severity fires, and the fire regimes associated with all but the lowest elevation dry forest type are well within what is expected of the natural fire regimes in the region (Arno *et al.*, 2000, Baker 2009).

The way salvage logging is carried out in both regions has also some particularities that are worth highlighting here. In our WMB study area, 80% of the forests are privately owned, so salvage logging is mainly intended to recoup the economic losses caused by fire. This means that the most usual postfire treatment consists in logging everything except the smallest snags and the trees that were not killed by fire, which results in large clearcuts. Selective logging is rare, and carried out almost entirely on publicly owned forests under the guise of forest restoration. In contrast, most NRM forests are public and managed by the US Forest Service. There, postfire salvage logging occurs in relatively small patches or management units (which average about 40 ha in extent). Managed burned forests in the NRM usually consist of a mix of unlogged patches, lightly logged patches, and clearcuts.

Table 1. Summary of the number of burned sites, their average size (in hectares) and the number of samples included in the study for each region. For further information see Table A2.1 in Annex 2.

Region	Fires	Fire size	Sample sizes[1] Total	UL	LS	HS
West Mediterranean Basin	23	648	384 transects	170	39	139
Northern Rocky Mountains	17	6,944	889 point counts	544	221	124

[1]*Postfire tratment: UL, unlogged; LS, light salvage logging; HS, heavy salvage logging.*

Data on birds and postfire management

We gathered data on the occurrence and abundance of birds in recently burned forests in association with independent monitoring projects. In WMB, birds were surveyed by line transects (Zozaya *et al.* 2011), counting birds along walking transects that were 15 min in duration. Each transect was separated from other transects by at least 150 m. In these

transects, birds were counted in three distance bands (0-25m, 25-50m and 50-100m). In NRM, surveys were carried out using 10-min point counts. Points were separated from one another by at least 250 m. At each point, observers registered the distance from the observer to the bird. In both surveys, birds flying over were not registered. Surveys were carried out in the breeding season (spring) in both cases. Taking into account the differences in methods, we used only the occurrence of each bird species detected within 50 m of the observer. The relatively short fixed distance allowed us to be confident that we detected most birds in the area immediately surrounding the transect or point count.

In both locations, we gathered information on the postfire management treatment performed at the sample level (transect or point count). Management possibilities were: 1) unlogged (UL), when the sample area was not logged; 2) light salvage logging (LS), when some snags were removed but the area still had an intermediate snag density and 3) heavy salvage logging (HS), when the area was basically clearcut and very few snags were left standing.

In order to focus on the most similar habitats and forms of disturbance, we chose only conifer forests that were burned by high-severity fires. Although surveys were conducted since the first year after fire in both locations, we used only those samples that were obtained in the third and fourth year after fire because of the lack of enough salvaged samples in the two first years after fire. This way we avoided comparing bird communities from samples derived from slightly different times after fire, which can be substantially different (Smucker *et al.* 2005). This resulted in a sample of 384 transects in 23 burned sites in WMB and 889 points in 17 sites in NRM (Table 1).

Species ranking

In order to compare different bird species' occurrence from both regions, we ranked all species detected according their theoretical dependence on snags or trees and, therefore, their expected sensitivity to salvage logging. We used known life-history traits to give each species a snag dependence value (SDV), an adaptation of the 'threat factor' ranking proposed by Imbeau *et al.* (2000). We used information from literature (Ehrlich *et al.* 1988; Snow & Perrins 1998; Estrada *et al.* 2004) and our own knowledge to assign three scores to each species, which depended on their habitat preference, foraging behavior, and nest location, respectively (Table 2). We obtained a SDV for each species

Table 2. Critera used to assign each species a score according to its habitat, nesting, and foraging requirements. Scores were then summed to calculate a snag dependence value (SDV).

Score	Habitat requirements	Nest location	Foraging behavior
0	generalists, farmland, grassland, shrubland, urban	ground, shrubs	ground, shrubs
1	open woods, forest edges, riparian bottomlands	snags and trees, including cavities	bark and canopy gleaners
2	forests	-	-

by summing these scores. We then ranked all the species detected in both regions from 4 to 0 according to their SDV (Table A2.3 in Annex 2). In order to ease data analysis and the subsequent interpretation of results, we reduced the number of SDV categories and grouped species with 4 and 3 SDV into a highly snag-dependent species (HSD) category, species with 2 and 1 SDV into an intermediate snag-dependent species (ISD) category, and species with 0 SDV into a not snag-dependent species (NSD) category.

Data analysis

In order to compare the response of HSD, ISD and NSD species to salvage logging, we calculated the number of those species in each sample (transect or point), and obtained the proportion of each class of species in each sample. We then analyzed the effect of the 3 different postfire management treatments on the proportionate contribution of these three species groups using Generalized Linear Mixed Models (GLMM), where we included the region, the interaction between regions (WMB and NRM) and the postfire treatment (UL, LS and HS), as fixed effects. Samples were not entirely independent because they were grouped in different fires, and some of them were surveyed twice, so we included the sample, nested within site, as a random factor in the model. Since response variables were proportions, we used a model with binomial error distribution and logit link. Furthermore, we also analyzed the response of the five most frequent species of each snag dependence group from each region. To do so we used the same random factors structure for each species, but including only postfire treatment as a fixed effect. We validated the significance of fixed effects with F tests with $p < 0.05$ judged to be statistically significant, and looked for differences between each factor category using multiple comparisons (Tukey tests). We performed all these analyses with SAS 9.1.

Results

We obtained information on the occurrence of 70 species in the WMB and of 80 in the NRM. The proportions of species with different degrees of snag dependence differed significantly between regions: both HSD species ($F_{1;279}=30.8$, $P<0.001$) and ISD species ($F_{1;279}=72.5$, $P<0.001$) represented a higher share of the bird community in NRM than in WMB. In contrast, the proportion of NSD species was higher in WMB ($F_{1;279}=206.2$, $P<0.001$).

Salvage logging affected species with different levels of snag dependence differently (Figure 2). While the proportion of HSD birds was negatively affected by more intense salvage logging, the proportion of ISD species did not show a clear trend. In contrast, the proportion of NSD species was not affected by salvage logging treatment in NRM, but was in WMB, where they represented a higher proportion of the bird community in highly logged areas than in unlogged ones.
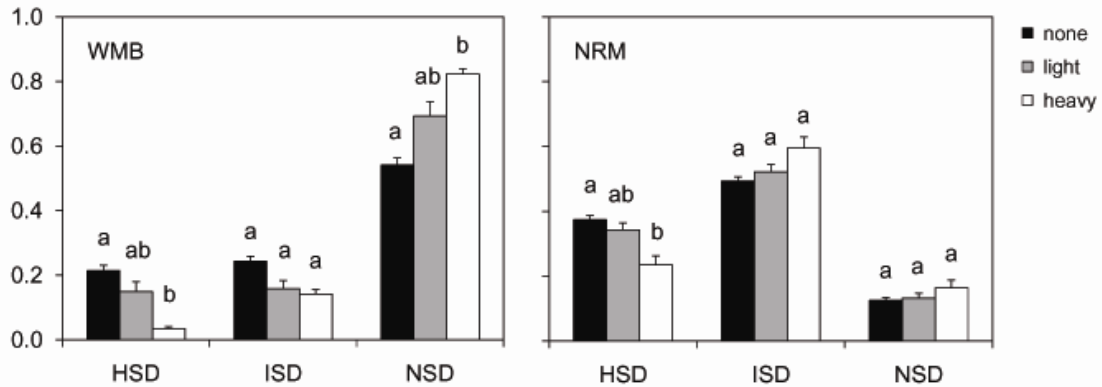
Figure 2. Proportion of highly snag-dependent species (HSD), intermediate snag-dependent species (ISD) and not snag-dependent species (NSD) in samples of burned forests that had not been logged (black bars), had been lightly logged (gray bars) or had been heavily logged (white bars), in the Western Mediterranean Basin (WMB) and in the Northern Rocky Mountains (NRM). Different letters above bars show significant differences (Tukey adjusted p<0.05) between postfire treatment categories, according to multiple comparisons tests.

The individual responses of the most frequently detected bird species can help us to understand these patterns of change in bird community composition (Figure 3; see Table A2.2 in Annex 2 for further details). Four of the five most commonly detected HSD species in WMB (Blue Tit, Crested Tit, Eurasian Chaffinch, and Short-toed Treecreeper), and two of the five most commonly detected HSD species in NRM (Yellow-rumped Warbler and Mountain Chickadee) showed a significant negative response to salvage logging. Similarly, the majority of ISD species did not show a strong response to salvage logging, but those that did (European Blackbird, European Goldfinch, and American Robin) were affected negatively by snag removal. In contrast, more NSD species showed positive than negative responses to salvage logging, and all of the significant responses were positive. However, the response of Lincoln's Sparrow, a commonly detected NSD species, could not be tested statistically since it was never detected in the HS treatment, suggesting a strong negative effect of salvage logging on its occurrence.

Discussion

In burned forests from regions as different as the Mediterranean Basin and the American Rocky Mountains, the drastic change in habitat structure following salvage logging produces a common pattern: the drastic decline of bird species that highly depend on snags. The abundance of most strictly forest birds from both regions (chickadees, tits, creepers, woodpeckers, nuthatches, arboreal warblers and finches) decreases markedly, as does the availability of snags that they need for nesting and foraging. For many bird species, the negative effect of salvage logging is not so drastic when performing a less intensive, intermediate salvage logging. Nevertheless, the
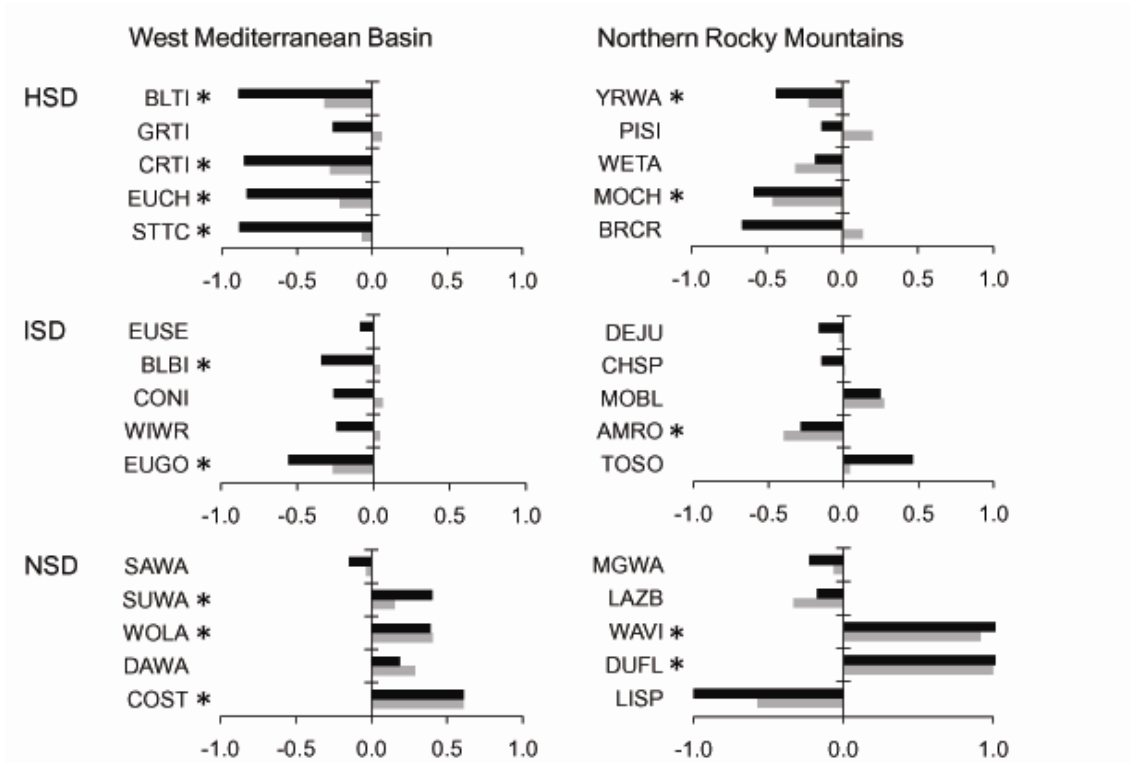
Figure 3. Response of five most common species[1] of each snag dependence category[2] to light and heavy salvage logging. Bars represent the proportion of samples with presence of the species in unlogged samples minus the proportion of samples with presence of the species in lightly logged (gray bars) or heavily logged (black bars), divided by the sum of both proportions. Asterisks indicate which species' occurrence showed significant differences (p<0.05) between postfire categories. No analysis could be performed with Lincoln's Sparrow (LISP) occurrence because it was not detected in any heavily logged sample.

[1]*BLTI: Blue Tit, GRTI: Great Tit, CRTI: Crested Tit, EUCH: Eurasian Chaffinch, STTC: Short-toed Treecreeper, EUSE: European Serin, BLBI: European Blackbird, CONI: Common Nightingale, WIWR: Winter Wren, EUGO: European Goldfinch, SAWA: Sardinian Warbler, SUWA: Subalpine Warbler, WOLA: Wood Lark, DAWA: Dartford Warbler, COST: Common Stonechat, YRWA: Yellow-rumped Warbler, PISI: Pine Siskin, WETA: Western Tanager, MOCH: Mountain Chickadee, BRCR: Brown Creeper, DEJU: Dark-eyed Junco, CHSP: Chipping Sparrow, MOBL: Mountain Bluebird, AMRO: American Robin, TOSO: Townsend's Solitaire, MGWA: McGillivray's Warber, LAZB: Lazuli Bunting, WAVI: Warbling Vireo, DUFL: Dusky Flycatcher, LISP: Lincoln's Sparrow.*

[2]*HSD: highly snag-dependent species, ISD: intermediate snag-dependent species, NSD: not snag-dependent species.*

occurrence or breeding success of several fire specialist bird species seems to depend on the presence of relatively densely stocked burned forests, and those bird species appear to be negatively affected even by incomplete salvage logging operations (Kotliar *et al.* 2002, Hutto & Gallo 2006, Saab *et al.* 2007, Hutto 2006, Hutto 2008).

The occurrence of bird species that were considered less dependent on the availability of snags (intermediate snag-dependent birds), and therefore could be less sensitive when facing salvage logging, did not show a significant response to any level of postfire salvage logging in either region or when considered as a whole. However, some species in this category showed negative responses to snag removal, suggesting that different

species can be sensitive to different thresholds of snag density. Even though their probabilities of occurrence may not have been affected by salvage logging, some other aspects of their biology may still have been negatively affected. For instance, even though the probability of occurrence of the Mountain Bluebird did not differ significantly between treatments in our study, this species' nest density was much lower in salvaged patches than that in unlogged ones due to the reduction in the number of cavities in logged areas (Hutto & Gallo 2006). Therefore, information about the occurrence of bird species, although being a useful metric for understanding the distribution and habitat preferences of species, should be complemented by other information related to the species' biology that may also be affected by salvage logging (e.g., issues related to breeding success).

Non-snag-dependent bird species showed different responses to salvage logging between regions. In the NRM they did not show a significant response as a whole group, while in the WMB their importance within the bird community increased in logged areas. That can be explained by differences in the biology of these species in each region. On one hand, both regions had a handful of shrub-dwelling birds (some warblers, shrikes) that, in general, did not show significant differences between unlogged and logged areas. These species are probably more linked to shrubland regeneration itself rather than to the presence of snags, although some of them, provided there is enough shrub cover, may prefer more open, logged areas (e.g., Subalpine Warbler, Warbling Vireo and Dusky Flycatcher). On the other hand, in the WMB there are many bird species that typically inhabit farmland and steppe-like habitats that colonize burned forests (Brotons *et al.* 2005, Pons & Bas 2005), and that generally benefit from salvage logging (Rost *et al.* 2010). Species such as larks, buntings or pipits are far more common in WMB burned areas than in NRM burned areas, which is probably a consequence of the different landscape configurations between regions. Open-habitat colonizers tend to occur in recently burned areas only if there are source populations close enough, because the probability of colonization is limited by dispersal constraints (Brotons *et al.* 2005, Brotons *et al.* 2008). Therefore, burned (and logged) areas located too far from source populations are less likely to be occupied than those areas located near those habitats supporting populations of potential colonizers. In the fine-grained Mediterranean landscape, forests are relatively close to farmland areas, and that can explain the difference in response of farmland birds between regions. On average, the distance between the burned forests we studied in the WMB and the nearest farmland area was just 180 m. In contrast, in the NRM, burned forests were two orders of magnitude farther away (on average, 17.8 km from prairies and pastures). Consequently, NRM burned and logged forests, that could be suitable for species from open habitats, were not as easily colonized as similar forests in WMB.

The spatial arrangement of logged stands in each region may also help to explain some unexpected effects of this disturbance, such as the weak response showed by some species that were *a priori* considered highly snag-dependent (e.g., Pine Siskin, Western Tanager). The distribution of salvage logging in small patches within NRM forests may

mask an otherwise detrimental effect, since these species may have been detected in logged areas primarily because of the close proximity of unlogged patches. Therefore, the negative effect of salvage logging could have been somewhat underestimated in the NRM.

The results of the comparison we carried out, although showing interesting common patterns between regions, also suggest that the impact of salvage logging cannot be considered equally severe in both regions, in the line of our initial predictions. Indeed, bird communities inhabiting burned forests are dominated by different kind of species in each region. In the NRM, highly snag-dependent birds constitute a higher proportion of the bird community (28 of 80 species, or 35%) than they do in the WMB (15 of 70 species, or 21%). Therefore, the impact of salvage logging can be greater in the NRM because it is detrimental to a higher proportion of the bird community that occurs in burned forests than in the WMB. Furthermore, salvage logging (either clearcut or intermediate) is clearly detrimental in the NRM to several fire specialists, like the Black-backed Woodpecker (*Picoides arcticus*) or the American Three-toed Woodpecker (*Picoides tridactylus*), which are restricted to dense and severely burned forests (Saab & Dudley 1998, Schwab *et al.* 2006, Hutto 2008). In the WMB, in contrast, none of the bird species can be considered to be fire specialists, and most species that are damaged by postfire salvage logging are common forest birds that have wide distributions across European forests, being neither rare nor threatened birds at a continental scale (Blondel & Aronson 1999, BirdLife International 2004). However, managers from certain areas of the Mediterranean Basin must be aware of the occurrence of narrowly distributed and endemic species, which may need to be preserved in the face of postfire salvage logging, like the Corsican Nuthatch (*Sitta whiteheadi*) (Moneglia *et al.* 2009). On the other hand, most Mediterranean open-habitat species that occur in burned forest patches show negative population trends in Europe due to the loss and deterioration of their habitat (Fuller *et al.* 1995, BirdLife International 2004). Therefore, the change in the cover type from forest to a more open habitat due to fire and salvage logging would be beneficial to them (Rost *et al.* 2010).

Conclusions

Postfire salvage logging appears to be strongly detrimental to species most closely tied to forest environments, a generalization that holds across two very different regions. We can expect that forest birds would be the most negatively affected by postfire logging in other parts of the world as well. However, the gravity of those negative effects depends on the particularities of each region, and therefore it seems advisable not to generalize management decisions on burned forests, which should take into account each region's characteristics and conservation priorities. Specifically, it can be considered to be more seriously detrimental to carry out salvage logging operations in a burned forest in NRM than in WMB, where the negatively affected species are generally neither threatened nor restricted to postfire conditions. In NRM, the

environment is still relatively pristine, and some species strongly depend on burned, unmanaged forest habitats. Thus, the ecological costs associated with postfire salvage logging appear to be much greater in NRM than they are in WMB. Therefore, it would be more advisable in NRM to concentrate timber harvesting operations in unburned forests located in and near the wildland-urban interface, where there is also a strong need to address issues of public safety.

Acknowledgements

References

Arno, S.F., Parsons, D.J. & Keane, R.E. 2000. Mixed-severity fire regimes in the northern Rocky Mountains: consequences of fire exclusion and options for the future. In: Cole, D.N., McCool, S.F., Borrie, W.T., O'Loughlin, J. (eds.). 2000. Wilderness science in a time of change. Proceedings RMRS-P-15-VOL-5. USDA Forest Service, Rocky Mountain Research Station, Ogden.

BirdLife International. 2004. Birds in Europe: population estimates, trends and conservation status.. BirdLife Conservation Series No. 12. BirdLife International, Wageningen.

Blondel, J. &Aronson, J. 1999. Biology and wildlife of the Mediterranean region. Oxford University Press, New York.

Brotons, L., Pons, P. &Herrando, S. 2005. Colonization of dynamic Mediterranean landscapes: where do birds come from after fire? Journal of Biogeography 32, 789-798.

Brotons, L., Herrando, S. &Pons, P. 2008. Wildfires and the expansion of threatened farmland birds: the ortolan bunting *Emberiza hortulana* in Mediterranean landscapes. Journal of Applied Ecology 45, 1059-1066.

Cahall, R.E. &Hayes, J.P., 2009. Influences of postfire salvage logging on forest birds in the Eastern Cascades, Oregon, USA. Forest Ecology and Management 257, 1119-1128.

Castro, J., Moreno-Rueda, G. & Hódar, J.H. 2010. Experimental test of post-fire management in pine forests: impact of salvage logging versus partial cutting and non-intervention on bird-species assemblages, Conservation Biology 24, 810-819.

Ehrlich, P.R., Dobkin, D.S. & Wheye, D. 1988. The Birder's Handbook. A Field Guide to the Natural History of North-American Birds. Simon & Schuster, Inc., New York.

Estrada, J., Pedrocchi, V., Brotons, L. & Herrando, S. (eds.) 2004. Atles dels ocells nidificants de Catalunya 1999-2002. Institut Català d'Ornitologia (ICO)/Lynx Edicions, Barcelona.

Fuller, R.J., Gregory, R.D., Gibbons, D.W., Marchant, J.H., Wilson, J.D., Baillie, S.R. & Carter, N. 1995. Population declines and range contractions among lowland farmland birds in Britain. Conservation Biology 9, 1425-1441.

González, J.R. & Pukkala, T. 2007. Characterization of forest fires in Catalonia (north-east Spain). European Journal of Forest Research 126, 421-429.

Hessburg, P.F. & Agee, J.K. 2003. An environmental narrative of Inland Northwest United States forests, 1800-200. Forest Ecology and Management 178, 23-59.

Hutto, R.L. 2006. Toward meaningful snag-management guidelines for postfire salvage logging in North American conifer forests. Conservation Biology 20, 984-993.

Hutto, R.L. 2008. The ecological importance of severe wildfires: some like it hot. Ecological Applications 18, 1827-1834.

Imbeau, L., Mönkkönen, M. & Desrochers, A. 2000. Long-term effects of forestry on birds of the Eastern Canadian boreal forests: a comparison with Fennoscandia. Conservation Biology 15, 1151-1162.

Izhaki, I. &Adar, M. 1997. The effects of postfire management on bird community succession. International Journal of Wildland Fire 7, 335-342.

Kotliar, N.B., Hejl, S.J., Hutto, R.L., Saab, V., Melcher, C.P. & McFadzen, M.E. 2002. Effects of fire and postfire salvage logging on avian communities in conifer dominated forests of the western United States. Studies in Avian Biology 25, 49-64.

Lindenmayer, D.B. & Noss, R.F. 2006. Salvage logging, ecosystem processes, and biodiversity conservation. Conservation Biology 20, 949-958.

Llimona, F., Matheu, E., & Prodon, R. 1993. Role of snag persistence and of tree regeneration in postfire bird successions: comparison of pine and oak forests in Montserrat (Catalonia, NE Spain). In: Trabaud, L., Prodon, R. (Eds.), Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Bruxelles. Pp. 315-331.

McIver, J.D. & Starr, L. (eds.). 2000. Environmental effects of postfire logging: literature review and annotated bibliography. General technical report PNW-GTR-486. USDA Forest Service, Pacific Northwest Research Station, Portland.

Morisette, J.L., Cobb, T.P., Brigham, R.M. & James, P.C., 2002. The response of boreal songbird communities to fire and postfire harvesting. Candian Journal of Forest Research 32, 2169-2183.

Moneglia, P., Besnard, A., Thibault, J-C. & Prodon, R. Habitat selection of the Corsican Nuthatch (Sitta whiteheadi) after a fire. Journal of Ornithology 150, 577-583.

Pausas, J.G. 2004. Changes in fire and climate in in the eastern Iberian Peninsula (Mediterranean basin). Climate Change 63, 337-350.

Pausas, J.G., Llovet, J., Rodrigo, A. & Vallejo, R. 2008. Are wildfires a disaster in the Mediterranean basin? –A review. International Journal of Wildland Fire 17, 713-723.

Pausas, J.G. & Keeley, J.E. 2009. A burning story: the role of fire in the history of life. BioScience 59, 593-601.

Pons, P. & Bas, J. 2005. Open-habitat birds in recently burned areas: the role of the fire extent and species' habitat breadth. Ardeola 52, 119-131.

Rost, J., Clavero, M., Bas, J.M. & Pons, P. 2010. Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests. Forest Ecology and Management 260, 79-86.

Saab, V.A., & J.G. Dudley. 1998. Responses of cavity nesting birds to stand-replacement fire and salvage logging in ponderosa pine/Douglas-fir forests of southwestern Idaho. Research Paper RMRS-RP-11. USDA Forest Service, Rocky Mountain Research Station, Ogden.

Saab, V., Russell, R.E. & Dudley, J.G. 2007. Nest densities of cavity-nesting birds in relation to postfire salvage logging and time since wildfire. Condor 109, 97-108.

Schwab, F.E., Simon, N.P.P., Stryde, S.W. & Forbes, G.J. 2006. Effects of postfire snag removal on breeding birds of western Labrador. Journal of Wildlife Management 70, 1464-1469.

Smucker, K.M., Hutto, R.L. & Steele, B.M. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15, 1535-1549.

Snow, D. & Perrins, C. (eds.). 1998. The complete Birds of the Western Palearctic on CD-ROM. Oxford University Press, Oxford.

Swanson, M. E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B., & Swanson, F.J. 2010. The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment, in press. DOI:10.1890/090157.

Zozaya, E.L., Brotons, L., Herrano, S., Pons, P., Rost, J. & Clavero, M. 2010. Monitoring bird community dynamics in Mediterranean landscapes affected by large wildfires. Ardeola 57, 33-50.

Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests

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Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests

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Abstract

Salvage logging is a common practice carried out in burned forests worldwide, and intended to mitigate the economic losses caused by wildfires. Logging implies an additional disturbance occurring shortly after fire, although its ecological effects can be somewhat mitigated by leaving wood debris on site. The composition of the bird community and its capacity to provide ecosystem services such as seed dispersal of fleshy-fruited plants have been shown to be affected by postfire logging. We assessed the effects of the habitat structure resulting from different postfire management practices on the bird community, in three burned pine forests in Catalonia (western Mediterranean). For this purpose, we focused on the group of species that is responsible for seed dispersal, a process which takes place primarily during the winter in the Mediterranean basin. In addition, we assessed microhabitat selection by seed disperser birds in such environments in relation to management practices. Our results showed a significant, positive relationship between the density of wood debris piles and the abundance of seed disperser birds. Furthermore, such piles were the preferred microhabitat of these species. This reveals an important effect of forest management on seed disperser birds, which is likely to affect the dynamics of bird-dependent seed dispersal. Thus, building wood debris piles can be a useful practice for the conservation of both the species and their ecosystem services, while also being compatible with timber harvesting.

Keywords: salvage logging, postfire management, winter bird community, frugivorous birds, seed dispersal, *Pinus halepensis*.

Introduction

Wildfires are one of the most important disturbances affecting ecological processes and economic interests in forest ecosystems worldwide (Blondel & Aronson 1999, Hessburg

& Agee 2003, FAO 2007). In many countries postfire salvage logging (i.e., the harvesting of burned timber for mainly commercial purposes) is the most extended forestry practice intended to compensate for the economic losses caused by wildfires (Hutto 2006; Lindenmayer *et al.* 2004). Salvage logging is also justified by other objectives, such as reducing the fuel load, enhancing future tree survival, preventing the proliferation of xylophagous insects or making it easier and safer for people to pass through the burned area or for future forest management work to be carried out (Ne'eman *et al.* 1995, Martínez-Sánchez *et al.* 1999, Lindenmayer & Noss 2006).

In the Mediterranean Basin, where there has been a long history of landscape management, fires often take place in lowland pine forests, most of which grow on abandoned agricultural land (Trabaud 1981). In this type of forest, after a fire, salvage logging is the most frequently used management method, because pine wood is still commercially profitable for use as boards until two years after a fire, and even later if ground down (e.g., to produce biomass fuel). In general terms, salvage logging involves a drastic habitat change —from a burned forest structure to open grassland and shrubland— a short time after the fire. However, on a smaller scale the characteristics of these new open habitats also depend on the interests and decisions of landowners and managers, and therefore on the ownership structure of each burned forest. When forests are divided into small privately-owned lots, as is often the case in Mediterranean areas, individual decisions lead to the coexistence of multiple management models in burned stands in close proximity. These strategies result in clearcuts (where wood debris may be completely removed, left on the ground or piled up), areas where isolated snags remain, and unlogged stands of partially burned pines. Furthermore, restoration measures such as log erosion barriers can be found in burned public forests, which are managed essentially for restoration purposes. The administrative authority concerned may also issue management recommendations, and grant subsidies to, or reach agreements with, local landowners, who may then delegate postfire management to public managers.

The habitat disturbance induced by postfire salvage logging affects animal communities, an issue that has sometimes been studied using birds as the focal group (Kotliar *et al.* 2002, Hutto & Gallo 2006, Cahall & Hayes 2009). It has been shown that salvage logging can have a greater impact than the fire itself on the forest bird community, including the substitution of forest specialists by open-land species (Izhaki 1993, Llimona *et al.* 1993, Morisette *et al.* 2002, Castro *et al.* 2010). Such changes are likely to affect frugivorous bird species, some of which are involved in important ecological processes such as seed dispersal of fleshy-fruited shrubs and climbers. In the Mediterranean, fleshy-fruited plants produce ripe fruit during the fall and winter (Mooney & Kummerow 1981, Jordano 1992), when various species of small and medium-sized birds disperse their seeds (Herrera 1984, Herrera 1998). In burned areas, early succession gives rise to a shrubland-like habitat (Trabaud & Lepart 1980), which hosts a diverse winter bird assemblage, including some species that attain high densities thanks to postfire fruit and seed production (Pons & Prodon 1996, Pons 2001, Tellería

76

et al. 2004). Izhaki & Adar (1997) showed that winter bird communities in eastern Mediterranean burned and logged forests were dominated by shrubland species, some of which were seed dispersers. However, the habitat and microhabitat use of these species under different scenarios of postfire management has not yet been studied in depth. Previous studies have reported that bird-dependent seed rain strongly relies on habitat structures that act as dispersal foci, such as isolated trees that act as perches (Guevara & Laborde, 1993; Pausas *et al.* 2006, Herrera & García 2009). After logging, wood debris piles and barriers built in a burned pine forest can also act as dispersal foci (Rost *et al.* 2009). Thus, the variability of postfire treatments after logging, which leads to the presence of different microhabitats on a small scale, may have important implications for bird-dependent seed rain, depending on the differential use that seed dispersers make of the resulting microhabitats.

The aims of this study are: (1) to investigate the effects of postfire management on winter bird communities in burned Mediterranean pine forests, (2) to determine whether certain management practices have positive effects on species richness and abundance of seed dispersers in winter, and (3) to analyze microhabitat selection by seed dispersers in order to describe patterns in their use of the microhabitats. If postfire management practices had an effect on seed dispersers distribution and microhabitat use, this would also affect seed rain and the regeneration of fleshy fruit-producing shrub or climber species. We took advantage of the availability in three study areas of examples of different postfire logging practices (ranging from completely logged areas to unlogged ones, and including different methods of managing wood debris) to investigate which of these practices may be more favorable to the presence of bird species involved in seed dispersal.

Methods

Study area

The study was carried out in three burned areas located in Catalonia, in the northeast of the Iberian Peninsula: Torroella de Montgrí (henceforth TM, 3º7' E, 42º3' N), Ventalló (VE, 3º2' E, 42º7' N) and Cistella (CI, 2º50' E, 42º17' N) (Figure 1). The maximum distance between any two areas was 30 km. All three areas have a meso-Mediterranean climate, limestone substrates, and altitudes ranging from 80 to 210 m above sea level. Before the fire, they featured a vegetation mosaic composed of Aleppo pine (*Pinus halepensis*) forests, shrubland and crops. TM burned in September 2004 and the fire affected 188 ha of pine forest; in VE and CI fires burned 613 and 170 ha of pine forest, respectively, in August 2006. All were severe fires that affected forest canopy and undergrowth, and as a result the vast majority of pines died and the shrub cover disappeared.
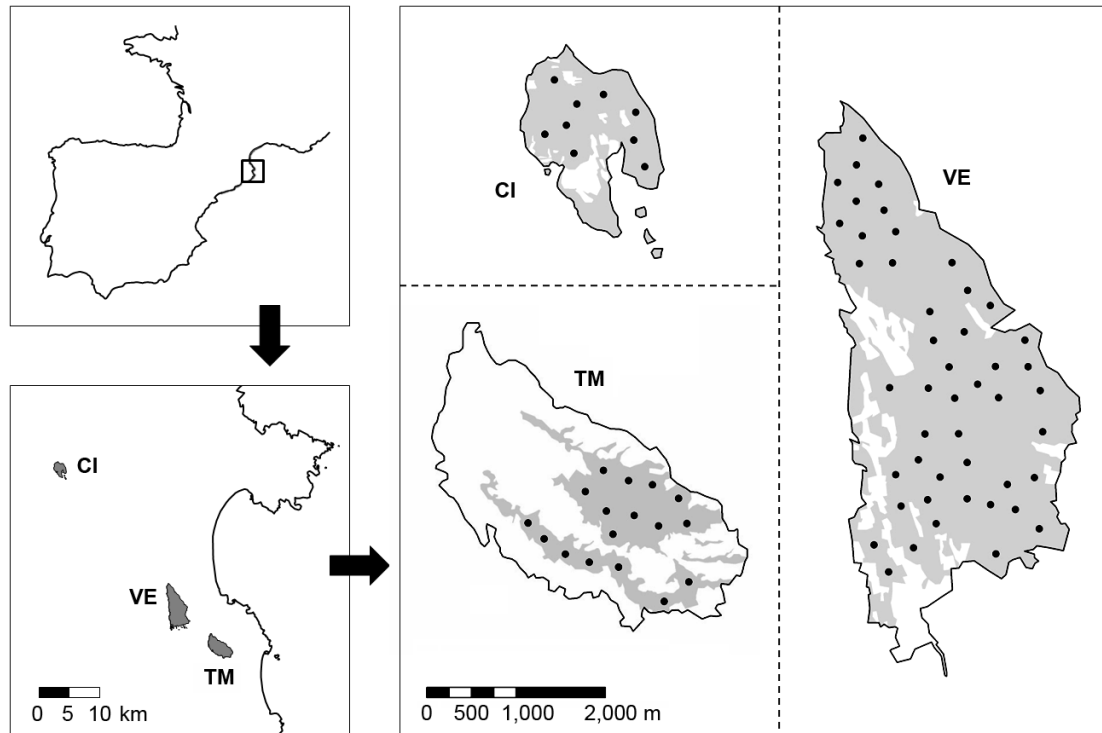
Figure 1. Location of the three study areas and distribution of the 76 stations (dots) within these areas (CI: Cistella; TM: Torroella de Montgrí; VE: Ventalló). Gray areas correspond to the former forests zones on which the study focused. Fire perimeters obtained from DMAH (2008).

Properties at VE and CI study areas are small and privately owned. In VE 96% of the area is in private ownership, with stands ranging in area from 0.5 to 5 ha and only a few stands larger than 15 ha, while in CI 100% of the area is in private ownership, with one stand of 80 ha and the rest ranging from 5 to 20 ha. In both areas salvage logging started during the first winter after the fires and was still happening when the fieldwork was conducted, affecting approximately 60% of the burned forest area in VE and about 75% in CI. The few forest patches where pines survived after the fires were left unlogged. In a few areas, non-profitable wood debris was heaped up in small, low piles. In TM, the predominance of public ownership (66% of the total area) led to patchy postfire management, including both logged and unlogged areas and restoration measures. In logged patches, however, a few standing snags were preserved (ca. 10 snags/ha) and log barriers were built up against runoff and erosion by piling trunks and branches along the contour lines (the sizes of the piles being 0.5-1 m high and 3-10 m long) (Rost *et al.*, 2009). By the time we conducted the fieldwork in TM all unlogged snags had fallen down, as had almost a third of those in VE and CI.

The study was conducted in winter 2008-09, the third winter after the fires in VE and CI and the fifth in the case of TM, when vegetation was in early succession. The regenerating undergrowth was composed of resprouter and seeder shrubs (*Quercus coccifera*, *Phillyrea angustifolia, Cistus* spp.) and herbs (*Brachypodium* spp.), with young Aleppo pine saplings in many stands. We concentrated all fieldwork in a single

month (January) in order to avoid as much as possible the changes in bird abundance that may happen along winter.

Bird sampling

We distributed 74 sampling stations of 100 m radius in the study areas, following a regular spatial distribution within the burned forests (48 stations in VE, 17 in TM and 9 in CI; Figure 1). When choosing the exact location of the stations we bore in mind to allow a minimum distance between station centers (300 m), to ensure good visibility, and to sample a homogeneous habitat (burned forest with the same management model). Recently logged areas (less than four weeks after logging) were not sampled in order to avoid the disturbance that logging work might have caused to the birds. Birds were counted using the point count method (Bibby *et al.* 1992) once at each station. All counts were performed by the same observer, conducted between one and four hours after dawn and under good weather conditions. All birds seen or heard within a 100 m radius during three consecutive five-minute periods were registered, but individuals seen just flying over were not counted, as they were not considered to be using the sampled habitat. In order to reduce the probability of double counts during the 15 minutes, we computed each species' abundance as the maximum number of individuals in any of the three five-minute periods. We initially separated bird observations in three distance bands from the observer (0-25 m, 25-50 m and 50-100 m) to account for detectability. However, the estimation of detection probabilities with Distance 5 program (Thomas *et al.* 2006) was only possible for the 42% most frequent species. For these species, bird density taking into account detection probability was highly correlated with the raw abundance (R^2=0.83). Moreover, at least 67% of detections of the species that accounted for 75% of the total abundance where auditory, so that birds were detected rather irrespectively of the cover structure. For these reasons we used raw abundances in all the analyses.

Habitat variables

Habitat variables related to vegetation recovery and postfire management were recorded at each station. Within a 20 m radius area centered on the station we estimated the relative area (in percentage) covered by: bare ground, vegetation up to 0.25 m, vegetation up to 0.5 m, vegetation up to 1 m, and vegetation higher than 1 m (excluding live trees), which all together totaled 100%. The density and cover of standing snags and live trees were measured in a 20 m radius area (if density was >100 trees/ha), 50 m radius area (between 10 and 100 trees/ha) or 100 m radius area (<10 trees/ha). The density (number/ha) and cover of wood debris piles or barriers in a 50 m radius area, and the cover of logging debris scattered on the ground in a 20 m radius area were considered as management variables. Cover measures were estimated by visual

comparison with the template used by Prodon & Lebreton (1981), except for scattered wood debris, for which we designed a specific template (Figure 2).
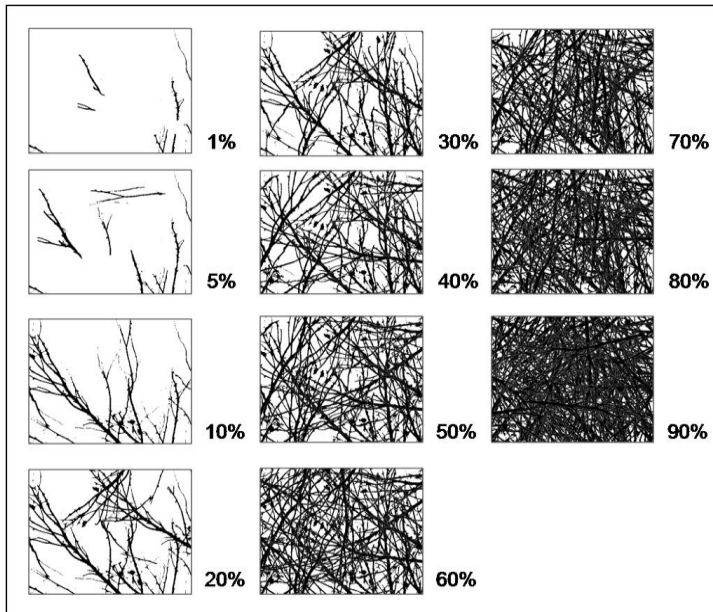
Figure 2. Template used to estimate the scattered wood debris cover on the ground.

Microhabitat selection by avian seed dispersers

As a result of the postfire management explained above, we defined five different microhabitat categories: live trees, standing snags, regenerated shrubs, wood debris piles, and bare ground. The availability of each microhabitat was estimated as its cover (in percentage) within the station. During point counts, we recorded the microhabitat, (only one in each case) in which individual birds were detected. For flushed out individuals, we recorded the microhabitat from which they had flown away. When this was not possible (e.g., in the case of wood piles among live shrubs and bare ground patches), the second site where the bird stopped was recorded, since the first movement might have been influenced by the presence of the observer. The behavior of seed disperser species of fleshy-fruited plants was also recorded, categorized as foraging, perching, sheltering or unknown (when a bird had been flushed out and not relocated). Only those species classified as major dispersers in Herrera (1995), that is, all species of the genus *Sylvia* and *Turdus* plus the European robin (*Erithacus rubecula*), were considered as seed dispersers in the analyses.

Data analyses

Multivariate analyses were used to explore the relationships between bird community and habitat structure. All species occurring in less than 5% of point counts were excluded. A redundancy analysis (RDA) was applied to the species and to the habitat and management variables matrices, including burned area as a covariate. A forward

selection of habitat variables was performed using Monte-Carlo permutation tests and including those variables that were significant at a $p<0.1$ level. In the RDA, two species occurring in 3-5% of the points were included as supplementary species (thus not influencing the analysis results). These were the European jay (*Garrulus glandarius*), because of its importance as oak acorns disperser, and the Southern grey shrike (*Lanius meridionalis*), because of the interest in its conservation in Europe (BirdLife International 2004). We used a linear method (RDA) instead of a unimodal approach because the length of the gradient of a DCA performed to the 20 species and 74 samples matrix was 2.75 SD, suggesting a mostly linear response of the species (ter Braak & Smilauer 1998). RDA and DCA were performed with CANOCO (ter Braak & Smilauer 1998). A Principal Components Analysis (PCA) with varimax rotation was performed to resume habitat variability in a reduced number of habitat gradients. The variables used were bare ground, all the live vegetation categories defined above, snag density, and live tree density. We interpreted the meaning of principal components using variables with factor loadings higher than 0.6, in absolute value. PCA gradients were kept to be used as predictor variables in subsequent analyses. We used Statistica 6 (Statsoft Inc. 2001) to perform this analysis.

Generalized linear mixed models (GLMM) were used to assess the importance of habitat-management variables on the following community variables: abundance and species richness of the overall bird community, and abundance and species richness of seed dispersers. Abundances of most common seed dispersers were also analyzed. We used first two PCA gradients, scattered wood debris cover and piles density as predictors. GLMM with Poisson error and link log were performed, including the locality as a random factor in order to control possible site-based differences. For each dependent variable, the minimum adequate model was selected following a manual backward selection procedure, retaining those predictors with p-values smaller than 0.05. GLMMs were performed with program R 2.9.2 (R Development Core Team 2009), using the lme4 software.

Microhabitat selection by seed disperser birds was analyzed for the overall seed dispersers group and for the three most abundant seed disperser species (Sardinian warbler [*Sylvia melanocephala*], Dartford warbler [*Sylvia undata*] and European robin) by comparing observed and expected bird observation frequencies in each microhabitat with chi-square tests. Expected frequencies were calculated from the availability of microhabitats in the stations where the species had been detected, after scaling these microhabitats' cover to 1. Behavior was not included in the analyses due to the excessive reduction of frequencies that would be involved by doing so, and was only represented graphically.

Results

In the point counts we recorded a total of 744 birds belonging to 38 species (Table 1). The most abundant species were the Sardinian warbler, the Dartford warbler, the European robin, the Hedge accentor (*Prunella modularis*) and the Chaffinch (*Fringilla coelebs*). The seed disperser group is composed by nine species (Table 1), which accounted for 58% of the total abundance, the three most abundant accounting for 48% of the total abundance.

RDA results showed that habitat variables explained 16% of the total variance of the species dataset, and the first two axes explained 87% of the species-environment relationships (eigenvalues: first axis = 0.08, second axis = 0.04). Covariates explained an additional 9% of the variance. The ordination of species along these two axes shows two main gradients, one related to regenerated vegetation structure and the other related to the density of snags and live trees (i.e., the logging intensity) (Figure 3). The first gradient separates species that require open areas with low vegetation cover, such as the Woodlark (*Lullula arborea*), from those that dwell in well-developed, high shrubland, such as the Sardinian warbler and the Blackcap. The second axis separates forest species that prefer high tree density, such as *Parus* tits or Chaffinches, from those that mainly select unwooded habitats in winter, which is the case of the Dartford warbler, the European robin or the Zitting cisticola (*Cisticola juncidis*).
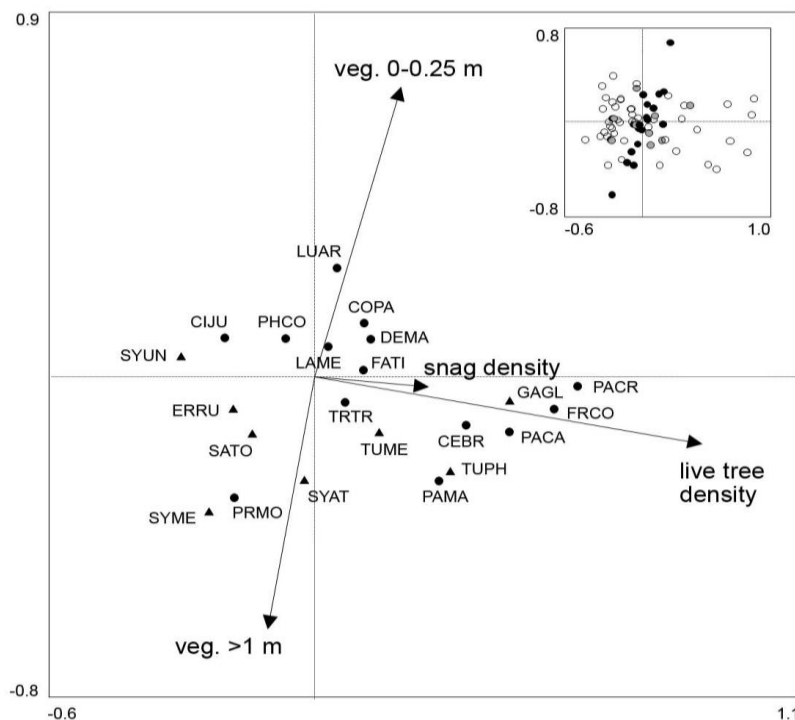
Figure 3. Biplot of the first two axes of the RDA, showing species and significant environmental variables (live vegetation up to 0.25 m, live vegetation higher than 1 m, snag density and live tree density). Triangle marks indicate seed disperser species. Species abbreviations are shown in Table 1. The inset graph (top left) shows the ordination of samples along the same axes, with dots representing the study areas (black-TM; gray-CI and white-VE).

82

Table 1. Species recorded in all 74 point counts with occurrence and total number of individuals. Asterisks show seed disperser species (Herrera 1984) of fleshy-fruit plants, and double asterisks show major disperser species (Herrera 1995). The horizontal bar separates those species considered in multivariate analyses from those not considered.

English name	Scientific name	Acronym	Occurrence	Sum of counts
Sardinian warbler **	*Sylvia melanocephala*	SYME	71	191
Dartford warbler **	*Sylvia undata*	SYUN	52	93
European robin **	*Erithacus rubecula*	ERRU	48	74
Hedge accentor	*Prunella modularis*	PRMO	33	44
Winter wren	*Troglodytes troglodytes*	TRTR	26	29
Chaffinch	*Fringilla coelebs*	FRCO	24	43
Great tit	*Parus major*	PAMA	20	30
Chiffchaff	*Phylloscopus collybita*	PHCO	16	16
Woodlark	*Lullula arborea*	LUAR	15	24
Song thrush **	*Turdus philomelos*	TUPH	13	28
Blackcap **	*Sylvia atricapilla*	SYAT	12	19
Blue tit	*Parus caeruleus*	PACA	12	17
Blackbird **	*Turdus merula*	TUME	11	14
Zitting cisticola	*Cisticola juncidis*	CIJU	10	12
Crested tit	*Parus cristatus*	PACR	8	10
Short-toed treecreeper	*Certhia brachydactyla*	CEBR	8	11
Great Spotted woodpecker	*Dendrocopos major*	DEMA	6	6
Stonechat *	*Saxicola torquata*	SATO	6	7
Common kestrel	*Falco tinnunculus*	FATI	4	4
Woodpigeon	*Columba palumbus*	COPA	4	12
European jay *	*Garrulus glandarius*	GAGL	3	4
Common buzzard	*Buteo buteo*	BUBU	2	2
Meadow pipit	*Anthus pratensis*	ANPR	2	2
Long-tailed tit	*Aegithalos caudatus*	AECA	2	5
Southern grey shrike	*Lanius meridionalis*	LAME	2	2
Goldfinch	*Carduelis carduelis*	CACD	2	6
Serin	*Serinus serinus*	SESE	2	11
Cirl bunting	*Emberiza cirlus*	EMCI	2	2
Peregrine falcon	*Falco peregrinus*	FAPE	1	1
Hoopoe	*Upupa epops*	UPEP	1	1
Green woodpecker	*Picus viridis*	PIVI	1	1
Alpine accentor	*Prunella collaris*	PRCO	1	3
Blue rock-thrush	*Monticola solitarius*	MOSO	1	1
Mistle thrush **	*Turdus viscivorus*	TUVI	1	1
Greenfinch	*Carduelis chloris*	CACH	1	2
Linnet	*Carduelis cannabina*	CACN	1	1
Red crossbill	*Loxia curvirostra*	LOCU	1	2
Corn bunting	*Miliaria calandra*	MICA	1	13

The first axis of the PCA explained 29% of total variance in habitat characteristics (eigenvalue = 2.04), and was positively related to vegetation up to 1 m and higher than 1 m, and negatively to bare ground and vegetation up to 0.25 m. The second axis explained a further 21% of the variance (eigenvalue=1.50), and was positively related to snag density and vegetation up to 0.5 m. The first factor thus represented undergrowth structures ranging from open areas to high shrublands, while the second ordered samples from clearcuts to areas with high snag density.

The overall bird abundance was higher in areas of low density of snags and live trees, while overall richness was not affected by any habitat-management predictor. The abundance of seed dispersers was positively related to pile density and shrub cover and negatively to snag density, and their species richness was not affected by any predictor. Shrub cover was also related to the abundance of the Sardinian warbler, the most common seed disperser. In addition to influence frugivores abundance, pile density favored the occurrence of the Dartford warbler, the European robin, and the Blackcap (*Sylvia atricapilla*). On the other hand, *Turdus* thrushes did not show any habitat-management preferences. See Table 2 for further details and significance values.

Both the seed disperser group and the three most abundant species (Sardinian warbler, Dartford warbler and European robin) used available habitat elements in a non-random fashion (Figure 5). As a group, seed dispersers were most frequently found on piles, resprouting shrubland and snags, using piles and snags more than would be expected from habitat element availability ($\chi^2=643.16$, p<0.01). In terms of individual species, the Sardinian Warbler, Dartford Warbler and European Robin prefered wood piles more than expected ($\chi^2=258.14$, p<0.01; $\chi^2=100.03$, p<0.01, and $\chi^2=173.10$, p<0.01, respectively). Behavior observations revealed that bird species used these elements for somewhat different purposes, with Sardinian and Dartford warblers using piles and shrubs basically for foraging, and European robins mostly for perching.

Discussion

Habitat heterogeneity in burned Mediterranean pine forests allows the coexistence of bird species with different habitat requirements during winter, and thus we found a mixture of open land, shrubland and forest birds in our study sites. Such habitat and bird community heterogeneity likely results from the management carried out in these areas, as well as from the regeneration of the low plant cover. We found that different management practices favored different bird species in winter. Indeed, managers decisions, such as the extent of snag logging and how wood debris is managed, that depend both on their objectives (harvest/restoration) and on the small size of lots in the area, greatly affect which bird species will occupy the area. Thus, strict open land birds may take profit both from fire, which creates open areas of low herbs and bare ground, and from logging, because of the mechanical damage to ligneous vegetation associated with logging operations, which hinders proper regeneration (Martínez-Sánchez *et al.* 1999, van Nieuwstadt *et al.* 2001, Lindenmayer & Noss, 2006). Shrubland birds dwell in the regenerated shrubs and bushes that have grown since fire and logging occurred, as well as in newly-built wood piles and barriers. In these clearcuts created by salvage logging, winter bird abundance is higher than in unlogged patches where snags have been retained. However, unlogged areas allow the persistence of some forest species that do not occur when snags are logged.

Table 2. Summary of GLMM selected models analyzing the effect of habitat and management variables (PCA1, PCA2, pile density and scattered wood debris cover) on the overall community abundance and richness, major disperser abundance and richness and most common major disperser species abundance, using locality as a random factor. Explained variance is shown as the variation in deviance between null and selected models, when both differed.

Response variable	Explained variance (%)	Predictors	Estimate	*p*
Overall community				
Abundance	4.3	PCA2	-0.090	0.016
Richness	0.0	-	-	-
Major seed dispersers				
Abundance	5.3	Pile density	0.009	0.017
		PCA1	0.141	0.005
		PCA2	-0.116	0.030
Richness	9.6	Pile density	0.009	0.088
Sardinian warbler	10.8	PCA1	0.200	0.006
Dartford warbler	4.7	Pile density	0.016	0.026
European robin	6.2	Pile density	0.023	0.003
Blackcap	15.0	Pile density	0.042	0.003
Blackbird	0.0	-	-	-
Song thrush	0.0	-	-	-
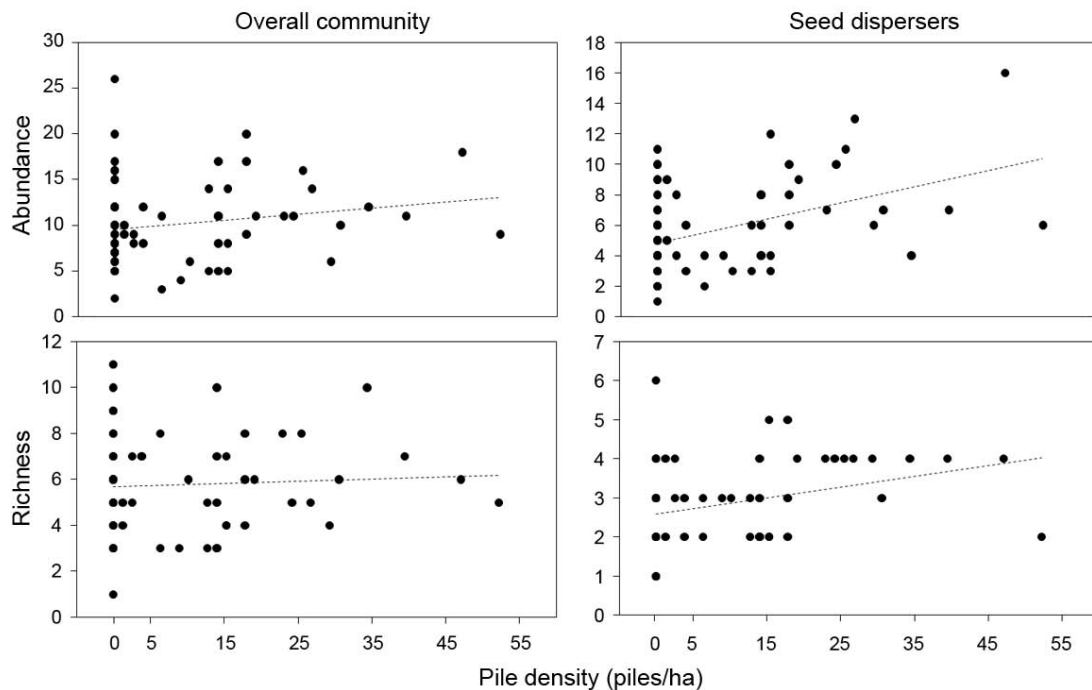
Figure 4. Effect of pile density on overall community and seed dispersers abundance and species richness. Only major seed dispersers have been considered (Herrera 1995). The dashed line represents the linear regression line.

Almost a quarter of all bird species and more than half of the individuals detected in the study area were seed dispersers, and among them, the Sardinian and Dartford warblers and the European robin were the most frequent species. In general, *Sylvia* warblers occurred mainly in bushy habitats without snags and trees. However, they showed some

85
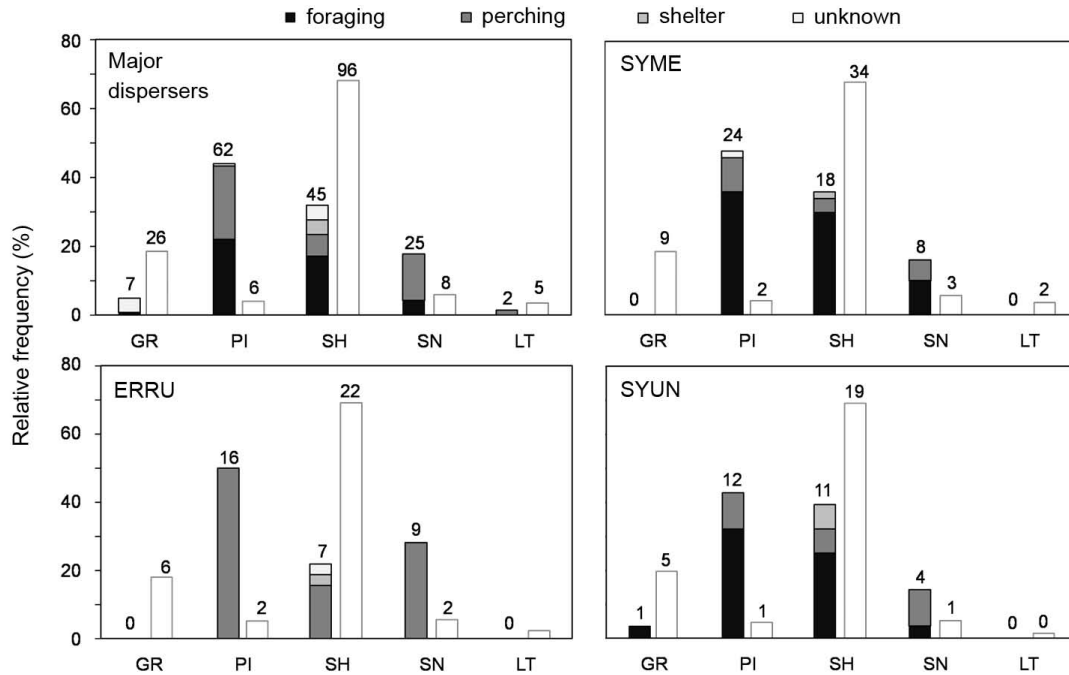
Figure 5. Relative frequencies of microhabitat use by all major disperser species, the Sardinian warbler (SYME), the European robin (ERRU) and the Dartford warbler (SYUN). For each habitat element (GR, ground; PI, pile; SH, shrub; SN, snag; LT, living tree), the left bar shows observed frequencies for four behavior categories, while the right white bar shows expected frequencies (for every graph, observed and expected frequencies add up 100%). The numbers above the bars show the number of observed or expected frequencies for each category.

differences in habitat preferences in accordance with their requirements (Shirihai *et al.* 2001), because Dartford warblers were found in lower shrubland than Sardinian warblers and Blackcaps. The European robin was found in similar habitats to those of warblers, since wintering individuals can occur in more open and drier habitats than breeding ones (Snow & Perrins 1998). On the other hand, *Turdus* thrushes were not closely linked to any specific habitat, but the Song thrush seemed more related to the presence of snags and live trees. By contrast, Castro *et al.* (2010) found that in Mediterranean montane pine forests, *Turdus* thrushes were the most abundant seed-disperser/frugivorous species in winter, the European robin was absent, and *Sylvia* warblers were uncommon. Salvage logging can thus have different effects on the seed dispersers group depending on the altitude, probably being more detrimental for seed dispersers in montane than in lowland forests.

The most remarkable result of our study is that seed dispersers are more abundant where piles of burned wood are built up in large quantities, being particularly important for Dartford warblers, European robins and Blackcaps. The vertical structure of these piles or barriers may give them the appearance of shrub patches and thus attract birds in early successional vegetation stages (Herrando *et al.* 2009), and seemed also to be important structures for species such as the European robin that are used to perching frequently when foraging, or as part of their territorial behavior. Furthermore, the occurrence of macro-arthropods in coarse wood debris (Grove 2002, Jabin *et al.* 2004, Varady-Szabo

& Buddle 2006) may increase the use of piles by insectivorous species. In addition, the preference for piles shown by frugivorous birds may have interesting ecological implications for the seed rain process in burned and managed areas, and support the role played by piles as seed dispersal foci (Rost *et al.* 2009). If dispersed seeds can germinate and survive under piles, a positive relationship between the abundance of fleshy-fruited plants and of frugivorous birds is likely, since piles may later host a greater abundance of such plants, and thus become important feeding points for frugivorous birds. However, the possible link between piles and fruit availability is a subject still to be studied. Apart from this, another way to deal with wood debris is to leave it scattered on the ground. Our results show that this management method did not have a big effect on the frugivorous guild.

Building wood debris piles in burned and logged areas may therefore be an important factor for fleshy-fruited plants, if afterwards these structures become points of high densities of dispersed and established plants. This could lead to a heterogeneous distribution of dispersed plant species within burned logged areas where piles have been built up. Seed disperser birds can bring seeds from unburned neighboring areas, where they may have eaten fruit, into burned areas (Rost *et al.* 2009). The arrival of migrant frugivores and the loss of territorial behavior of local ones during winter could allow longer dispersal distances than in spring-summer, when seed dispersal by birds takes place only at short distances (i.e., mostly below 100 meters) (Jordano *et al.* 2006). Therefore, birds attracted to burned areas by the presence of wood debris piles could facilitate the colonization of these areas by the dispersed plant species, with possible positive consequences for the recovery of the vegetation after fire.

The implications, suggested here, of the effect of management practices on seed disperser birds and on the seed dispersal process, are also likely to apply to forests altered by other harvesting practices. Common forest management measures in the Mediterranean, such as undergrowth clearing and tree thinning, intended to produce open forest stands for pasture, to maximize timber production or to reduce the risk of fire by eliminating the fuel load, are likely to cause such changes. Some existing studies of these management techniques and bird communities in the Mediterranean region (Camprodon & Brotons 2006, De La Montaña *et al.* 2006) found that some major seed dispersers (Sardinian warbler, Blackcap, Garden warbler *Sylvia borin* and Blackbird) disappear or decline locally when shrub clearing and thinning occur. This may mean a disruption in the seed dispersal process in such managed areas because of the lack of dispersers, negatively affecting bird-dispersed plant species. In such situations, piling up cut branches and other woody debris could therefore be seen as palliative measures, because they may provide shrubland-dwelling seed disperser birds with some resources that may favor their site tenacity.

Management recommendations

Our results suggest that the abundance and distribution of bird species in burned lowland pine forests in the western Mediterranean during the winter is strongly influenced by the implementation of postfire forest management practices. As shown in our results, if managers and landowners decide to harvest the burned pines, the fate of wood debris (branches and non-profitable logs) is not a trivial issue for birds, because it affects the abundance of major seed disperser species. Our recommendation, since scattered wood debris has no effect on birds abundance, is to collect and heap it up in piles of at least 0.5 m high. Around 15-25 piles/ha could be an appropriate density to allow the persistence of seed disperser birds in winter. Furthermore, wood piles, if found in an appropriate density, also have a positive effect on the species richness and abundance of breeding bird species (Herrando *et al.* 2009), on the abundance of some species of rodents (Haim & Izhaki 1994, Manning & Edge 2008), and even on the European rabbit (Catalán *et al.* 2008, Joan Real pers. comm.), a keystone of Mediterranean ecosystems. In most cases, the main purpose of building up wood debris piles is to reduce soil loss by erosion (Robichaud *et al.* 2000), although it is not clear that they are useful for that purpose (Raftoyannis & Spanos 1998, Marquès & Mora 1998). Nevertheless, their value for birds and other fauna would justify their inclusion in postfire management plans aiming to restore the habitat or introduce environmentally-friendly measures. Such practices seem feasible in public pine forests where postfire management is not so closely linked to economic imperatives, and could be made a requirement to obtain public subsidies for the management of private forests. Including the building up of wood debris piles in postfire management guidelines could thus become an important factor in helping to conserve both species and animal-dependent ecological services in the Mediterranean region. Finally, dialogue with landowners of burned forests is a necessary step to find ways to make economic objectives compatible with the benefits of biodiversity, thus facilitating the maintenance of important ecological functions such as bird-dependent seed dispersal processes.

Acknowledgements

References

Bibby, C.J., Burgess, N.D. & Hill, D.A. 1992. Bird census techniques. Cambridge University Press, Cambridge.

BirdLife International. 2004. Birds in Europe: population estimates, trends and conservation status. BirdLife Conservation Series No. 12. BirdLife International, Wageningen.

Blondel, J. & Aronson, J. 1999. Biology and Wildlife of the Mediterranean Region. Oxford University Press, New York.

ter Braak, C. J. F. & Smilauer, P. 1998. CANOCO reference manual and User's Guide to Canoco for Windows. Software for Canonical Community Ordination (Version 4). Microcomputer Power, Ithaca, New York.

Cahall, R.E. & Hayes, J.P. 2009. Influences of postfire salvage logging on forest birds in the Eastern Cascades, Oregon, USA. Forest Ecology and Management 257, 1119-1128.

Camprodon, J., & Brotons, L. 2006. Effects of undergrowth clearing on the bird communities of the Northwestern Mediterranean Coppice Holm oak forests. Forest Ecology and Management 221, 72-82.

Castro, J., Moreno-Rueda, G. & Hódar, J.H. 2010. Experimental test of post-fire management in pine forests: impact of salvage logging versus partial cutting and non-intervention on bird-species assemblages, Conservation Biology 24, 810-819.

Catalán, I., Rodríguez-Hidalgo, P. & Tortosa F.S. 2008. Is habitat management an effective tool for wild rabbit (*Oryctolagus cuniculus*) population reinforcement? European Journal of Wildlife Research 54, 449-453.

De La Montaña, E., Rey-Benayas, J.M. & Carrascal, L.M. 2006. Response of bird communities to silvicultural thinning of Mediterranean maquis. Journal of Applied Ecology 43, 651-659.

DMAH (Departament de Medi Ambient i Habitatge, Generalitat de Catalunya). 2008. Incendis 1986-2008 - Medi Ambient i Habitatge. http://mediambient.gencat.cat.

FAO. 2007. State of the World's Forests 2007. FAO, Rome.

Grove, S.J. 2002. Saproxylic insect ecology and the sustainable management of forests. Annual Review of Ecology, Evolution and Systematics 33, 1–23.

Guevara, S. & Laborde, J., 1993. Monitoring seed dispersal at isolated standing trees in tropical pastures –consequences for local species availability. Vegetatio 108, 319–338

Haim, A. & Izhaki, I. 1994. Changes in rodent community during recovery from fire: relevance to conservation. Biodiversity and Conservation 3, 573-585.

Herrando, S., Brotons, L., Guallar, S., Sales, S. & Pons, P. 2009. Postfire forest management and Mediterranean birds: the importance of logging remnants. Biodiversity and Conservation 18, 2153-2164.

Herrera, C.M. 1984. A study of avian frugivores, bird-dispersed plants, and their interaction in Mediterranean scrublands. Ecological Monographs 54, 1-23.

Herrera, C.M. 1995. Plant-vertebrate dispersal systems in the Mediterranean: Ecological, Evolutionary and Historical Determinants. Annual Review of Ecology, Evolution and Systematics 26, 705-726.

Herrera, C.M. 1998. Long-term dynamics of Mediterranean frugivorous birds and fleshy fruits: a 12-year study. Ecological Monographs 68, 511-538.

Herrera, J.M. & García, D., 2009. The role of remnant trees in seed dispersal through the matrix: being alone is not always so sad. Biological Conservation 142, 149–158.

Hessburg, P.F. & Agee, J.K. 2003. An environmental narrative of Inland Northwest United States forests, 1800–2000. Forest Ecology and Management 178, 23-59.

Hutto, R.L. 2006. Toward Meaningful Snag-Management Guidelines for Postfire Salvage Logging in North American Conifer Forests. Conservation Biology 20, 984-993.

Hutto, R.L. & Gallo, S.M. 2006. The effects of postfire salvage logging on cavity-nesting birds. The Condor 108, 817-831.

Izhaki, I. 1993. The resilience to fire of passerine birds in an East-Mediterranean pine forest on Mount Carmel, Israel: the effects of postfire management. In: Trabaud, L. & Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Bruxelles. Pp. 303-314.

Izhaki, I. & Adar, M. 1997. The effects of postfire management on bird community succession. International Journal of Wildland Fire 7, 335-342.

Jabin, M., Mohr, D., Kappes, H. & Topp, W. 2004. Influence of deadwood on density of soil macro-arthropods in a managed oak-beech forest. Forest Ecology and Management 194, 61-69.

Jordano, P. 1992. Fruits and frugivory. In: Fenner, M. (ed.). Seeds. The Ecology of Regeneration in Plant Communities. C.A.B. International, Wallingford. Pp. 125-166.

Jordano, P., García, C., Godoy, J.A. & García-Castaño, J.L. 2006. Differential contribution of frugivores to complex seed dispersal patterns. Proceedings of the National Academy of Sciences 104, 3278-3282.

Kotliar, N.B., Hejl, S.J, Hutto, R.L., Saab, V., Melcher, C.P. & McFadzen, M.E. 2002. Effects of fire and postfire salvage logging on avian communities in conifer-dominated forests of the western United States. In: George, T.L. & Dobkin, D.S. (eds.). Effects of habitat fragmentation on birds in western landscapes: contrasts with paradigms from the eastern United States. Studies in Avian Biology No. 25. Cooper Ornithological Society, Camarillo. Pp. 49-64.

Lindenmayer, D.B., Foster, D.R., Franklin, J.F., Hunter, M.L., Noss, R.F., Schmiegelow, F.A. & Perry, D. 2004. Salvage harvesting policies after natural disturbance. Science 303, 1303.

Lindenmayer, D.B. & Noss, R.F. 2006. Salvage Logging, Ecosystem Processes and Biodiversity Conservation. Conservation Biology 20, 949-958.

Llimona, F., Matheu, E. & Prodon, R. 1993. Role of snag persistence and of tree regeneration in postfire bird successions: comparison of pine and oak forests in Montserrat (Catalonia, NE Spain). In Trabaud, L. and Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Bruxelles. Pp. 315-331.

Manning, J.A. & Edge, W.D. 2008. Small mammal responses to fine woody debris and forest fuel reduction in southwest Oregon. Journal of Wildlife Management 72, 625-632.

Marquès, M.A. & Mora, E. 1998. Effects on erosion of two postfire management practices: clearcutting versus non-intervention. Soil & Tillage Research 45, 433-439.

Martínez-Sánchez, J.J., Ferrandis, P., de las Heras, J. & Herranz, J.M. 1999. Effect of burnt wood removal on the natural regeneration of Pinus halepensis after fire in a pine forest in Tus valley (SE Spain). Forest Ecology and Management 123, 1-10.

Mooney, R.E. & Kummerow, J. 1981. Phenological development of plants in mediterranean-climate regions. In: Di Castri, F., Goodall, D.W. & Specht, R.L. (eds.), Ecosystems of the World 11. Mediterranean Type Shrublands. Elsevier, Amsterdam. Pp. 303-307.

Morisette, J.L., Cobb, T.P., Brigham, R.M. & James, P.C. 2002. The response of boreal songbird communities to fire and postfire harvesting. Canadian Journal of Forest Research 32, 2169-2183.

Ne'eman, G., Lahav, H. & Izhaki, I. 1995. Recovery of vegetation in a natural east Mediterranean pine forest on Mount Carmel, Israel as affected by management strategies. Forest Ecology and Management 75, 17-26.

Pausas, J.G., Bonet, A., Maestre, F.T. & Climent, A., 2006. The role of the perch effect on the nucleation process in Mediterranean semi-arid oldfields. Acta Oecologica 29, 346–352.

Pons, P. 2001. The wintering of migrant Dunnocks *Prunella modularis* in two Mediterranean habitats after fire. Bird Study 48, 68-75.

Pons, P. & Prodon, R. 1996. Short term temporal patterns in a Mediterranean shrubland bird community after fire. Acta Oecologica 17: 29-41.

Prodon, R. & Lebreton, J. D. 1981. Breeding avifauna of a Mediterranean succession: the holm oak and cork oak series in the eastern Pyrenees, 1. Analysis and modelling of the structure gradient. Oikos 37, 21-38.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Raftoyannis, Y. & Spanos, I. 2005. Evaluation of log and branch barriers as postfire rehabilitation treatments in a Mediterranean pine forest in Greece. International Journal of Wildland Fire 14, 183–188.

Robichaud, P.R., Beyers, J.L. & Neary, D.G. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. USDA Forest Service, Fort Collins.

Rost, J., Pons, P. & Bas, J.M. 2009. Can salvage logging affect seed dispersal by birds into burned forests? Acta Oecologica 35, 763-768.

Shirihai, H., Gargallo, G. & Helbig, A., 2001. *Sylvia* warblers: identification, taxonomy and phylogeny of the genus *Sylvia*. Helm, London.

Snow, D. & Perrins, C. (eds.). 1998. The complete Birds of the Western Palearctic on CD-ROM. Oxford University Press, Oxford.

StatSoft, Inc. 2001. STATISTICA for Windows [Computer program manual]. StatSoft, Inc., Tulsa.

Tellería, J.L., Ramírez, A. & Pérez-Tris, J. 2004. Conservation of seed-dispersing migrant birds in Mediterranean habitats: Shedding light on patterns to preserve processes. Biological Conservation 124, 493–502.

Thomas, L., Laake, J.L., Strindberg, S., Marques, F.F.C., Buckland, S.T., Borchers, D.L., Anderson, D.R., Burnham, K.P., Hedley, S.L., Pollard, J.H., Bishop, J.R.B. & Marques, T.A. 2006. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, St. Andrews.

Trabaud, L. 1981. Man and fire: impacts on Mediterranean vegetation. In: di Castri, F., Goodall, D.W., Pecht, R.L. (eds). Mediterranean-type shrublands. Ecosystems of the world 11. Elsevier, Amsterdam. Pp. 523–537.

Trabaud, L. & Lepart, J., 1980. Diversity and stability in garrigue ecosystems after fire. Vegetatio 43, 49-57.

Van Nieuwstadt, M.G., Shiel, D. & Kartawinata, K. 2001. The ecological consequences of logging in the burned forests of east Kalimantan, Indonesia. Conservation Biology 15, 1183–1186.

Varady-Szabo, H. & Buddle, C.M. 2006. On the relationships between ground-dwelling spider (Araneae) assemblages and dead wood in a northern sugar maple forest. Biodiversity and Conservation 15, 4119-4141.

Capítol 4

Can salvage logging affect seed dispersal by birds into burned forests?

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Can salvage logging affect seed dispersal by birds into burned forests?

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Abstract

The recovery of vegetation in Mediterranean ecosystems after wildfire is mostly a result of direct regeneration, since the same species existing before the fire regenerate on-site by seeding or resprouting. However, the possibility of plant colonization by dispersal of seeds from unburned areas remains poorly studied. We addressed the role of the frugivorous, bird-dependent seed dispersal (seed rain) of fleshy-fruited plants in a burned and managed forest in the second winter after a fire, before on-site fruit production had begun. We also assessed the effect on seed rain of different microhabitats resulting from salvage logging (erosion barriers, standing snags, open areas), as well as the microhabitats of unlogged patches and an unburned control forest, taking account of the importance of perches as seed rain sites. We found considerable seed rain by birds in the burned area. Seeds, mostly from Olive trees *Olea europaea* and Evergreen pistaches *Pistacia lentiscus*, belonged to plants fruiting only in surrounding unburned areas. Seed rain was heterogeneous, and depended on microhabitat, with the highest seed density in the unburned control forest but closely followed by the wood piles of erosion barriers. In contrast, very low densities were found under perches of standing snags. Furthermore, frugivorous bird richness seemed to be higher in the erosion barriers than elsewhere. Our results highlight the importance of this specific post-fire management in bird-dependent seed rain and also may suggest a consequent heterogeneous distribution of fleshy-fruited plants in burned and managed areas. However, there needs to be more study of the establishment success of dispersed seeds before an accurate assessment can be made of the role of bird-mediated seed dispersal in post-fire regeneration.

Keywords: wildfire, frugivorous birds, seed rain, post-fire management, colonization, microhabitat.

Introduction

Mediterranean plant communities have been defined as resilient to fire (Lavorel 1999) because of the widespread occurrence of persistence organs and the strong recovery

95

capacity found in vegetation. After a disturbance, plants regenerate by two main strategies, resprouting and seed germination, depending on each species' life traits (Pausas 2001, Buhk *et al.* 2007). Resprouters are plants that shoot buds from undamaged lignotuber, epicormic tissues, rhizomes, roots, etc. Seeders have dry fruits with small seeds which germinate from the seed bank after heat stimulus caused by fire (Verdú 2000, Keeley & Fotheringham 2000). Both regeneration strategies allow most Mediterranean plant communities to regenerate directly in a process that has been called autosuccession, that is, the fast recovery of a plant community with the same species pool that it had before the disturbance (Trabaud & Lepart 1980). However, not all plant communities are capable of direct regeneration (Rodrigo *et al.* 2004) and even if they were, propagule colonization could play an unnoticed role in post-disturbance succession.

Seed dispersal, by different agents, is the natural process that is most important in allowing plant colonization of disturbed habitats (Neeman & Izhaki 1996). Directed dispersal by animals towards locations with particular characteristics is fundamental for successful seedling establishment and recruitment (Howe & Smallwood 1982, Schupp & Fuentes 1995). In the Mediterranean Basin the fruit of most fleshy-fruited plants ripens in autumn-winter, when it can be consumed by carnivore mammals and by wintering and resident frugivorous birds. These endozoochorous vertebrates digest the fruit pulp but defecate the viable seeds at some distance from the mother plant. In general, fleshy-fruited plants have big, short-life seeds that are killed by heat, and resprout after fire (Pausas & Verdú 2005). During the first months following the fire, these plants dedicate their strength to vegetative growth, and most of them do not bear fruit until more than a year after the fire. Throughout this initial period, the endozoochorous dispersal of their seeds from unburned habitats to burned areas could play a role in regeneration. However, this process remains to be studied (but see Cowling *et al.* 1997).

The pattern and composition of the seed rain can determine the regenerative potential of fleshy-fruited plants dispersed by endozoochorous vertebrates (Howe & Smallwood 1982, Schupp & Fuentes 1995). Factors that influence the composition of the seed rain include the density, distribution and fecundity of the seed sources in the vicinity, the spatial pattern of dispersal by the agents involved, and landscape features, such as perch availability for birds, that influence seed deposition (Murray 1988, Izhaki *et al.* 1991). Post-fire management, such as salvage logging operations, may take place after a forest fire. Logging alters the habitat use of birds, including frugivorous species (Llimona *et al.* 1993, Izhaki & Adar 1997, Herrando *et al.* in press), and could therefore affect the quantity and spatial distribution of dispersed seeds belonging to fleshy-fruited plants. Hence, the removal of perches (surviving trees, snags and burned branches) could be a critical issue, because of the importance of perches in seed deposition patterns originating from bird transport (Guevara & Laborde 1993, Pausas *et al.* 2006, Herrera & García 2009).

We took advantage of a large area of Mediterranean forest and shrubland that had recently been severely burned to investigate the magnitude of seed dispersal of fleshy-fruited plants by birds (i.e. seed rain). Since the plants of the burned area had not yet come to fruition during the study, all the collected seeds originated in unburned areas, which made it possible to assess genuine seed colonization in a disturbed area. A second aim of this study was to analyse the effects of post-fire management on seed rain by birds. In this connection, we predicted that the removal of perches would negatively affect the magnitude of seed rain in burned and managed forests.

Methods

Study area and post-fire management

The study was carried out in the Santa Caterina valley (3º7' E, 42º3' N), in the Montgrí massif (Catalonia, in the north-east of the Iberian Peninsula). This is a limestone area with a meso-Mediterranean climate. In September 2004 a wildfire burned 627 ha of Aleppo pine forest (*Pinus halepensis*), shrubland, rocky outcrops and crops. It was a severe fire which affected the forest canopy and undergrowth and caused widespread tree mortality. The study was conducted in the second winter after the fire (winter 2006-07) in a burned area of c. 90 ha, and in an unburned control area at a distance of 2 km (Figure 1). The vegetation in the burned area was at an early regeneration stage and consisted of low shrubs (Kermes oak *Quercus coccifera*, Evergreen pistache *Pistacia lentiscus*, rock roses *Cistus* spp.), climbing plants (Rough bindweed *Smilax aspera*, Wild madder *Rubia peregrina* and Wild asparagus *Asparagus acutifolius*) and grass (*Brachypodium retusum*). Within the burned study area only one patch of 2 ha was unburned. This patch was the nearest source of seeds (of native fleshy-fruit shrubs, some planted olive trees *Olea europaea var. europaea* and European hackberries *Celtis australis*). We knew this because we visually inspected the burned area and verified that fleshy-fruited plants were not fruiting in the regenerating shrubland. The unburned control area was an Aleppo pine forest unburned for the latest 50 years and with well-developed undergrowth.

Most of the burned study area (c. 70 ha) was managed for timber harvesting. It was public property so homogeneous salvage logging, consisting of felling almost all pine snags but leaving 5-10 standing snags/ha, was carried out. Non-commercially profitable logs and branches were piled along the contour lines in order to avoid soil erosion in this rugged terrain. The size of these barriers ranged between 0.5 and 1 m high and 3-10 m long (Figure 2a). The density of barriers was very high in some areas, reaching 100 barriers/ha (Figure 2b). Open areas, covered by regenerated low shrub vegetation, were found between the barriers. The rest of the burned study area (patches totalling up to 15-20 ha) was not managed and pine snags (200-400 snags/ha) remained standing until they fell naturally in following years. Finally, the unburned control Aleppo pine forest was unmanaged.
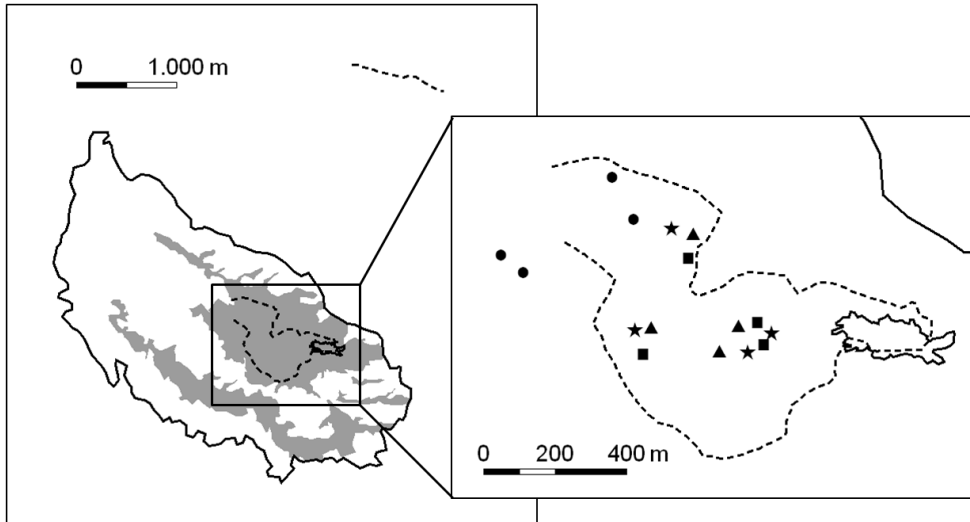
Figure 1. Left: Area burned by the 2004 wildfire (the solid line is the fire perimeter) and location of the bird census transects in burned and unburned areas (dashed lines); the extent of pine forest is shown in grey and the square shows the area enlarged at right. Right: location of the bird transect and seed trap sites in the burned study area (circles: unlogged; stars: barriers; squares: open areas; triangles: isolated trees; the last three microhabitats were located in logged areas) The two-hectare unburned patch is also drawn.
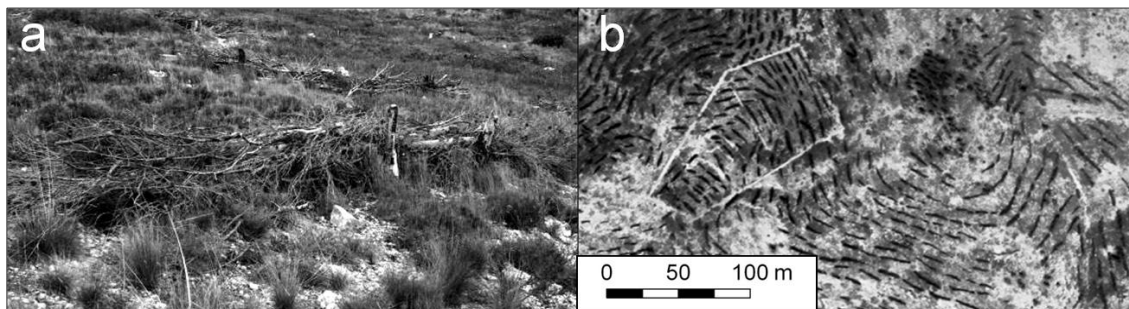
Figure 2. Log and branch erosion barriers in the study area. (a) A typical barrier; (b) Aerial photograph showing the distribution of barriers (dark lines).

Frugivorous bird sampling

Bird censuses were conducted once a month, from November to February, in order to record frugivorous species that could potentially disperse the seeds of fleshy-fruited plants. A line transect (Bibby *et al.* 1992) 3.3 km long was carried out within the burned study area (Figure 1). Approximately 40% of the total length of the transect passed through unlogged areas, while the rest was in salvage logged areas. A second transect (900 m long) was carried out in the unburned control area. Bird species occurrence was recorded within a band 40 m wide on each side of the pathway. We defined three microhabitats in the logged area: isolated snags, barriers and open areas. The location of birds among the three microhabitats was recorded in order to assess the microhabitat use of frugivorous species. We also classified bird species as fleshy-fruit plant dispersers in line with the available information for the Mediterranean region in autumn-winter (Herrera 1984, 1998).

98

Management effects on seed rain

Dispersion of fleshy-fruited plant seeds by birds was assessed in the five microhabitats resulting from management or its absence: unburned control forest, burned unlogged forest, and isolated snags, barriers and open areas in the burned logged forest. Seeds were collected using seed traps installed under perches from where birds could defecate. Perches consisted of living trees, snags, isolated snags, barriers or resprouting sprouts, depending on the microhabitat. Seed traps consisted of circular 0.07 m^2 plastic trays with a height of 4 cm. Each tray was covered by a 0.7 cm^2 wire mesh to prevent seed predation by rodents and birds. A sampling unit was made up of five of these traps put together and separated by 0.5 m from each other below the same perch. Four sampling units were set up in each microhabitat. Sampling units of the different microhabitats were interspersed in the available area. However, as management was not under our control, unlogged sites could not be interspersed with the other three microhabitats in the burned area (Figure 1). Sampling units, within and among microhabitats, were located at different distances from the nearest seed-source areas (unburned pinewood patch and burned area perimeter), ranging between 200 and 2,500 m from these sources. The seed traps were installed in November and checked monthly until early March. Once the seeds had been collected, they were counted and identified using a reference seed collection. We then obtained the dispersed seed density per plant species in each sampling unit, by dividing the number of seeds collected by the sampling unit area.

Data analysis

We used general linear mixed models (GLMM) to assess whether the distance from the seed source area affects the density of seeds found in traps. To this aim we used only the microhabitats where seed rain had to come from a distant seed source, thereby, excluding the unburned control. Seed density was the dependent variable in the GLMM, the distances between each replicate and the seed sources were the explicative variables and the four microhabitats from the burned area were the random factor, in order to control their possible effect on seed rain. We tested the seed results for the native shrubland species (Evergreen pistache, Wild asparagus, Rough bindweed and Wild madder) together since we assumed they had a common potential origin (i.e. the unburned patch and the unburned shrubland adjacent to the burned area perimeter). We separately tested the seed results of the Olive trees with a potential origin in the unburned patch and in the groves to the south and on the west side of the burned area. In both cases we used, alternately, the minimum distance to the nearest potential source (usually the unburned patch) and the mean distance between the two/three nearest potential sources as the explicative variable in the model.

We also performed ANOVA tests to detect differences in seed dispersal between microhabitats and plant species. For a first analysis, we used the seeds collected for all the plant species together, and we conducted two more, separate, tests for the Olive tree

99

and the Evergreen pistache, the two most dispersed species. In all cases, seed density was the dependent variable and microhabitats were the categorical predictors. Post-hoc differences were tested with Tukey tests. Seed density was square root transformed in order to fit the normality assumption. Statistical tests were performed with STATISTICA 8 software (StatSoft Inc. 2001). In all cases we considered $P < 0.05$ as being statistically significant.

Results

Frugivorous birds

We found 25 bird species in transect censuses, eight of which have a frugivorous diet in autumn-winter. These species were both forest species (found in the control and burned sites) and open-habitat ones: European Robin (*Erithacus rubecula*), Black redstart (*Phoenicurus ochruros*), Stonechat (*Saxicola torquata*), Blackbird (*Turdus merula*), Song thrush (*Turdus philomelos*), Sardinian warbler (*Sylvia melanocephala*), Dartford warbler (*Sylvia undata*) and Blackcap (*Sylvia atricapilla*) (Table 1). Frugivorous bird richness reached its highest value in wood debris barriers (66.7% of the total number of bird species), followed by open areas (46.2%), unburned control (35.3%), snags in the unlogged area (30.0%) and isolated snags (15.4%). The eight frugivorous species were observed in the barriers, with *Sylvia* warblers being especially frequent, and six of them were found in the open area around the barriers (although less frequently), or perched on the isolated snags of the logged area (occasionally). On the other hand, *Turdus* thrushes were the only species recorded in unlogged areas.

Table 1. Frequency of occurrence of frugivorous bird species in the five microhabitats (CO: unburned control, UN: burned unlogged, IS: isolated snags, OP: open area and BA: barriers) obtained from the four monthly censuses.

	Microhabitat				
Species	**CO**	**UN**	**IS**	**OP**	**BA**
European Robin *Erithacus rubecula*	0.75				0.25
Black redstart *Phoenicurus ochruros*				0.50	0.50
Stonechat *Saxicola torquata*				0.75	0.75
Blackcap *Sylvia atricapilla*	0.50				0.75
Sardinian warbler *Sylvia melanocephala*	1.00		0.25	0.25	1.00
Dartford warbler *Sylvia undata*			0.25	0.50	1.00
Blackbird *Turdus merula*	1.00	0.50		0.25	0.50
Song thrush *Turdus philomelos*	0.75	0.75			0.25
Total richness	5	2	2	5	8

Seed rain and management effect

We collected a total of 135 seeds of six fleshy-fruit plant species: Olive tree, N=83; Evergreen pistache, N=37; Wild asparagus, N=10; Rough bindweed, N=2; European hackberry, N=1; and Wild madder, N=1. The highest densities of Olive tree seeds were

found in barriers, whereas Evergreen pistache was much more dispersed in the unburned control forest (Table 2).

The distance from the potential seed sources to the sampling sites had no significant effect on seed dispersal (Table 3) either for native plant species or for Olive trees. However, microhabitat affects significantly the seed rain of olives. The ANOVA revealed significant differences in seed dispersal between microhabitats ($F_{4,15} = 5.18$, $P < 0.01$). The barrier and unburned control microhabitats had the highest values of seed density, reaching more than ten seeds/m^2 in the four-month sampling period. The lowest values were found in the burned unlogged, open area, and in especially, isolated snag microhabitats (Figure 3). However, post-hoc testing only showed that unburned control and barrier microhabitats were significantly different from the isolated snag microhabitats. Nevertheless, the effect of microhabitat depended on plant species: being significant for both Olive tree ($F_{4,15} = 3.21$, $P = 0.04$) and Evergreen pistache ($F_{4,15} = 4.60$, $P = 0.01$). Post-hoc comparisons show that significant differences existed between barriers and selective logging microhabitats for the Olive tree, and between the unburned control and burned unlogged, selective logging and open areas microhabitats for the Evergreen pistache (Figure 3).

Table 2. Mean density (in seeds/m^2 ± standard error) of fleshy fruit plant seeds collected in traps per microhabitat (CO: unburned control, UN: burned unlogged, IS: isolated snags, OP: open area and BA: barriers).

	CO	UN	IS	OP	BA
Olive tree *Olea europaea*	13.44 ± 7.0	7.78 ± 4.9	0.71 ± 0.7	8.49 ± 3.1	19.10 ± 7.9
Evergreen pistache *Pistacia lentiscus*	16.98 ± 4.8	2.12 ± 1.3	2.12 ± 0.7	3.54 ± 2.7	3.54 ± 0.7
Wild asparagus *Asparagus acutifolius*	4.24 ± 1.8	2.12 ± 1.3	-	-	2.83 ± 2.0
Rough bindweed *Smilax aspera*	0.71 ± 0.7	-	-	-	0.71 ± 0.7
European hackberry *Celtis australis*	-	-	-	-	0.71 ± 0.7
Wild madder *Rubia peregrina*	-	-	-	-	0.71 ± 0.7
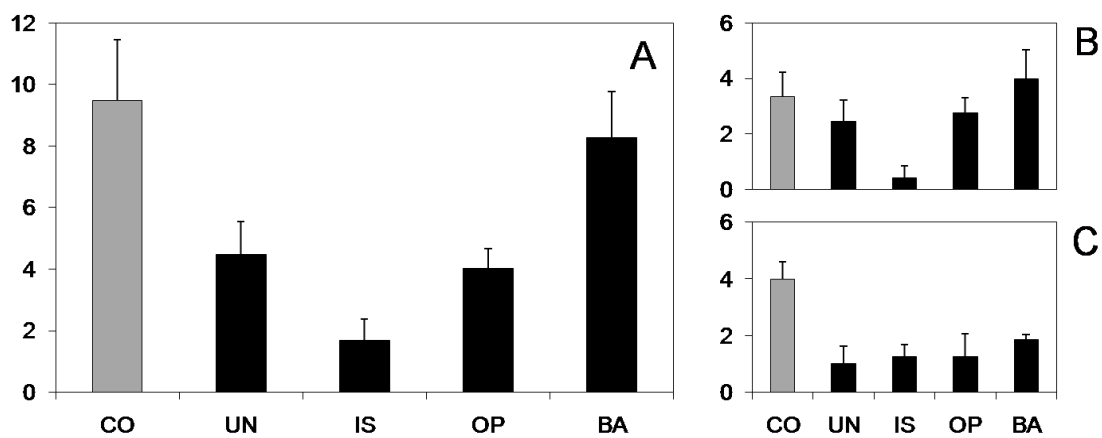
Figure 3. Mean density of collected seeds per square meter (square root transformed) in each microhabitat. A: sum of all plant species; B: Olive tree seeds; C: Evergreen pistache seeds. Black bars correspond to burned sites (CO: unburned control, UN: burned unlogged, IS: isolated snags, OP: open area and BA: barriers). Error bars indicate standard error.

101

Table 3. Linear regression analysis between seed density and distance from the nearest seed source, using microhabitats as the random factor.

	SS	df	F	*p*
Native shrub species				
Nearest source				
Distance	8.79	1	0.66	0.432
Microhabitat	76.45	3	1.93	0.183
Average 2 nearest sources				
Distance	1.10	1	0.08	0.784
Microhabitat	70.61	3	1.69	0.226
Olea europaea				
Nearest source				
Distance	0.31	1	2.67	0.130
Microhabitat	2.40	3	6.86	0.007
Average 3 nearest sources				
Distance	0.26	1	2.17	0.169
Microhabitat	2.25	3	6.19	0.010

Discussion

Our results show that there exists a bird-dependent seed rain in Mediterranean burned and managed forests, originating from seed sources in unburned areas. Furthermore, the quantity of seeds coming into a burned area is not very different in magnitude from the seed rain measured in nearby unburned forests. However, the density of seeds found in burned logged forests was heterogeneous, which suggests that habitat and microhabitat do have an important effect on seed rain (Debussche & Isenmann 1994, Kollmann & Pirl 1995, Milton *et al.* 1997).

Seed dispersal of fleshy-fruited plants can be affected by the different occupancy of microhabitats by frugivorous birds. The highest seed density in the burned area was concentrated in wood barriers, which may be attributable to the frequency of frugivorous bird species in this microhabitat. Bird species recorded in barriers usually occur in low vegetation, such as open shrublands or forest undergrowth, and wood barriers may favour their presence since these structures are the main perches in the short term after fire, when shrub regeneration is still very limited. Birds may actively use wood barriers for shelter, roosting, perching and even as feeding points. Even without fruiting plants, coarse woody debris can be a source of macro-arthropods in winter (Jabin *et al.* 2004, Varady-Szabo & Buddle 2006), and the majority of the frugivorous species we found also have an insectivore component in their diet (Snow & Perrins 1998). Consequently, the higher use of the barriers by frugivorous-insectivorous birds could be the cause of the higher seed dispersion rate in this microhabitat. Besides, such widely different microhabitats as unlogged burned patches and open areas both show evidence of a similar seed rain. The intermediate seed density found in completely logged areas could be due to the close proximity of barriers, from where frugivorous birds can occasionally visit the surrounding area. This edge effect at small scale may increase the seed rain in those open areas. In contrast, the similar seed density found in unlogged burned patches can be explained by the fact that only thrushes were found in such areas. On the other hand, few bird species used the isolated snags resulting from

selective logging, probably because of the lack of a suitable canopy cover, a snag density that is too low for thrushes, and because warblers do not usually perch on the highest branches. Therefore, contrary to our predictions, remaining isolated snags do not seem to act as dispersal sites for frugivorous bird depositions in such areas. Nevertheless, we showed that the effect of microhabitat on seed rain varies with the seed species. Such differences may stem from the local availability of each plant fruits, since Evergreen pistache fruits are probably much more abundant in unburned control, where seed rain has found to be higher, than Olive tree fruits, where it is not a common plant.

The Olive tree, a cultivated variety grown in nearby groves, was the species for which more seeds were found. Despite its high seed dispersal rate in the study area, its post-dispersal seeding capability seems very limited in natural conditions, and it is not considered an invasive species. Large thrushes (Song Thrush and Blackbird) are probably responsible for the dispersal of these seeds, since the gape size of birds limits the swallowing of large olive fruits (Rey *et al.* 1997). Hence, smaller frugivores should be considered as pulp peckers (Herrera 1984, Rey & Gutiérrez 1996) and not seed dispersers of commercial olive trees. On the other hand, the smaller fruits of native species may have been dispersed either by thrushes or by other major frugivores (mainly *Sylvia* warblers and the European Robin, see Herrera 1995). However, despite thrushes being scarcer than other species in barriers, that is where commercial olive seed rain reached its highest value (Table 2). Furthermore, the fact that we could not find an effect of distance from seed sources on seed density suggests that bird dispersers can move considerable distances between the burned and the unburned area. Jordano *et al.* (2006) found that small-sized frugivores (warblers) rarely disperse seeds at more than 100 m, and the nearest seed source was at c. 200 m to its nearest sampling unit. This also suggests that the majority of seed dispersal in our area is attributable to thrushes, the largest bird dispersers found in the burned area.

Bird depositions produce a directed, non-random seed rain (Murray 1988, Izhaki *et al.* 1991, Herrera *et al.* 1994). Nevertheless, the number of dispersed seeds is only a quantitative factor (Schupp 1993) since plant recruitment depends also on the microhabitat's suitability for seedling emergence and establishment (the nurse-plant effect), including factors such as competition, light intensity, temperature, soil conditions, rodent predation and herbivore damage (Tewksbury & Lloyd 2001, Kollmann 2000, García & Obeso 2003). Although wood barriers only represent a 5-10% of a burned and managed forest area, and therefore the absolute seed rain is probably not higher there than over other more extensive microhabitats, barriers may be more suitable because of their similarity to shrubs, where the germination of certain species is higher (García-Fayos & Verdú 1998). However, seed predation by rodents tends to be more intense under shrubs than in open areas (Kollmann 1995, Hulme 1996), and wood piles can benefit certain rodents (Manning & Edge 2008). Therefore, further investigation is needed to evaluate whether bird-dispersed seeds can successfully establish themselves in those microhabitats so that they can act as recruitment sites for

fleshy-fruited plants. If they could, the differential seed rain between microhabitats would lead to a heterogeneous distribution of fleshy-fruited plants in burned and managed areas in which a fine-grained mosaic of structures (isolated snags, wood barriers and open areas derived from logging) occurs. In contrast to our study area, small, privately-owned forest properties are the rule in Catalonia, a pattern that promotes patchy management. And finally, there needs to be further investigation into frugivorous bird distribution in autumn-winter in burned areas, in order to determine their habitat and microhabitat relationships more accurately, and to assess the effects of post-fire management on these species.

Acknowledgements

References

Bibby, C.J., Burgess, N.D. & Hill, D.A. 1992. Bird census techniques. Cambridge University Press, Cambridge.

Buhk, C., Meyne, A. & Jentsch, A. 2007. The challenge of plant regeneration after fire in the Mediterranean Basin: scientific gaps in our knowledge on plant strategies and evolution of traits. Plant Ecology 192, 1-19.

Cowling, R.M., Kirkwood, D., Midgley, J.J. & Pierce, S.M. 1997. Invasion and persistence of bird-dispersed, subtropical thicket and forest species in fire-prone coastal fynbos. Journal of Vegetation Science 8, 475-488.

Snow, D. & Perrins, C. (eds.). 1998. The complete Birds of the Western Palearctic on CD-ROM. Oxford University Press, Oxford.

Debussche, M. & Isenmann, P. 1994. Bird-dispersed seed rain and seedling establishment in patchy Mediterranean vegetation. Oikos 69, 414-426.

deWolfe, V.G., Santi, P.M., Ey, J. & Gartner J.E. 2008. Effective mitigation of debris flows at Lemon Dam, La Plata County, Colorado. Geomorphology 96, 366-377.

García, D. & Obeso, J.R. 2003. Facilitation by herbivore-mediated nurse plants in a threatened tree, *Taxus baccata*: local effects and landscape level consistency. Ecography 26, 739-750.

García-Fayos, P. & Verdú, M. 1998. Soil seed bank, factors controlling germination and establishment of a Mediterranean shrub: *Pistacia lentiscus* L. Acta Oecologica 19, 357-366.

Guevara, S. & Laborde, J. 1993. Monitoring seed dispersal at isolated standing trees in tropical pastures - consequences for local species availability. Vegetatio 108, 319-338.

Herrando, S., Brotons, L., Guallar, S., Sales, S. & Pons, P. 2009. Postfire forest management and Mediterranean birds: the importance of logging remnants. Biodiversity and Conservation 18, 2153-2164.

Herrera, C.M. 1984. A study of avian frugivores, bird-dispersed plants, and their interaction in Mediterranean scrublands. Ecological Monographs 54, 1-23.

Herrera, C.M. 1995. Plant-vertebrate seed dispersal systems in the Mediterranean: ecological, evolutionary, and historical determinants. Annual Review of Ecology Evolution and Systematics 26, 705-727

Herrera, C.M. 1998. Long-term dynamics of Mediterranean frugivorous birds and fleshy fruits: a 12-year study. Ecological Monographs 68, 511-538.

Herrera, C.M., Jordano, P., López-Soria, L. & Amat, J.A. 1994. Recruitment of a mast-fruiting, bird-dispersed tree: bridging frugivore activity and seedling establishment. Ecological Monographs 64, 315-344.

Herrera, J.M. & García, D. 2009. The role of remnant trees in seed dispersal through the matrix: Being alone is not always so sad. Biological Conservation 142, 149-158.

Howe, H.F. & Smallwood, J. 1982. Ecology of seed dispersal. Annual Review of Ecology Evolution and Systematics 13, 201-228.

Hulme, P.E. 1996. Natural regeneration of yew (*Taxus baccata* L.): microsite, seed or herbivore limitation? Journal of Ecology 84, 853-861.

Izhaki, I., Walton, P.B. & Safriel, U.N. 1991. Seed shadows generated by frugivorous birds in an eastern Mediterranean scrub. Journal of Ecology 79, 575-590

Izhaki, I. & Adar, M. 1997. The effects of post-fire management on bird community succession. International Journal of Wildland Fire, 7, 335-342.

Jabin, M., Mohr, D., Kappes, H. & Topp, W. 2004. Influence of deadwood on density of soil macro-arthropods in a managed oak-beech forest. Forest Ecology and Management 194, 61-69.

Jordano, P., García, C., Godoy, J.A. & García-Castaño, J.L. 2006. Differential contribution of frugivores to complex seed dispersal patterns. Proceedings of the National Academy of Sciences of the United States of America 104, 3278-3282.

Keeley, J. E. & Fotheringham, C.J. 2000. Role of fire in regeneration from seed. In: Fenner, M. (ed.). Seeds: the ecology of regeneration in plant communities. C.A.B. International, Wallingford. Pp 311-330.

Kollmann, J. & Pirl, M. 1995. Spatial pattern of seed rain of fleshy-fruited plants in a scrubland-grassland transition. Acta Oecologica 16, 313-329.

Kollmann, J. 1995. Regeneration window for fleshy-fruited plants during scrub development on abandoned grassland. Écoscience 2, 213-222.

Kollmann, J. 2000. Dispersal of fleshy-fruit species: a matter of spatial scale? Perspectives in Plant Ecology, Evolution and Systematics 3, 29-51.

Lavorel, S. 1999. Ecological diversity and resilience of Mediterranean vegetation to disturbance. Diversity and Distributions 5, 3-13.

Llimona, F., Matheu, E. & Prodon, R. 1993. Role of snag persistence and of tree regeneration in postfire bird successions: comparison of pine and oak forests in Montserrat (Catalonia, NE Spain). In: Trabaud, L. & Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Bruxelles. Pp. 315-331.

Malanson, G. P. 1987. Diversity, stability and resilience: effect of fire regimes. In: Trabaud L. (ed.). The role of fire in ecological systems. SPB Academic Publishing, The Hague. Pp. 49-63

Manning, J.A. & Edge, W.D. 2008. Small mammal responses to fine woody debris and forest fuel reduction in southwest Oregon. Journal of Wildlife Management 72, 625-632.

Milton, S.J., Dean, W.R.J. & Klotz, S. 1997. Thicket formation in abandoned fruit orchards: processes and implications for the conservation of semi-dry grasslands in Central Germany. Biodiversity and Conservation 6, 275-290.

Murray, K.G. 1988. Avian seed dispersal of three neotropical gap-dependent plants. Ecological Monographs 58, 271-298.

Neeman G. & Izhaki, I. 1996. Colonization in an abandoned East-Mediterranean vineyard. Journal of Vegetation Science 7, 465-472.

Pausas, J.G. 2001. Resprouting vs. seeding –a Mediterranean perspective. Oikos 94, 193-194.

Pausas, J.G. & Verdú, M. 2005. Plant persistence traits in fire-prone ecosystems of the Mediterranean basin: a phylogenetic approach. Oikos 109, 196-202.

Pausas, J.G., Bonet, A., Maestre, F. T. & Climent, A. 2006. The role of the perch effect on the nucleation process in Mediterranean semi-arid oldfields. Acta Oecologica 29, 346-352.

Raftoyannis, Y. & Spanos, I. 2005. Evaluation of log and branch barriers as post-fire rehabilitation treatments in a Mediterranean pine forest in Greece. International Journal of Wildland Fire 14, 183-188.

Rey, P.J. & Gutiérrez, J.E. 1996. Pecking of olives by frugivorous birds. A shift in feeding behaviour to overcome gape limitation. Journal of Avian Biology 27, 327-333,

Rey, P.J., Gutiérrez, J.E., Alcántara, J. & Valera, F. 1997. Fruit size in wild olives: implication for avian seed dispersal. Functional Ecology 11, 611-618.

Rodrigo, A., Retana, J. & Pico, X. 2004. Direct regeneration is not the only response of Mediterranean forests to large fires. Ecology 85, 716-729.

Schupp, E.W. 1993. Quantity, quality and the effectiveness of seed dispersal by animals. Vegetatio 107/108, 15-29

Schupp, E.W. & Fuentes, M. 1995. Spatial patterns of seed dispersal and the unification of plant population ecology. Écoscience 2, 267-275.

StatSoft, Inc. 2001. STATISTICA for Windows [Computer program manual]. StatSoft Inc., Tulsa.

Tewksbury, J.J. & Lloyd, J.D. 2001. Positive interactions under nurse-plants: spatial scale, stress gradients and benefactor size. Oecologia 127, 425-434

Trabaud, L. & Lepart, J. 1980. Diversity and stability in garrigue ecosystems after fire. Vegetatio 43, 49-57.

Varady-Szabo, H. & Buddle, C.M. 2006. On the relationships between ground-dwelling spider (Araneae) assemblages and dead wood in a northern sugar maple forest. Biodiversity and Conservation 15, 4119-4141.

Verdú, M. 2000. Ecological and evolutionary differences between Mediterranean seeders and resprouters. Journal of Vegetation Science 11, 265-268.

Capítol 5

The importance of piling wood debris on the distribution of bird-dispersed plants in burned and logged Mediterranean pine forests

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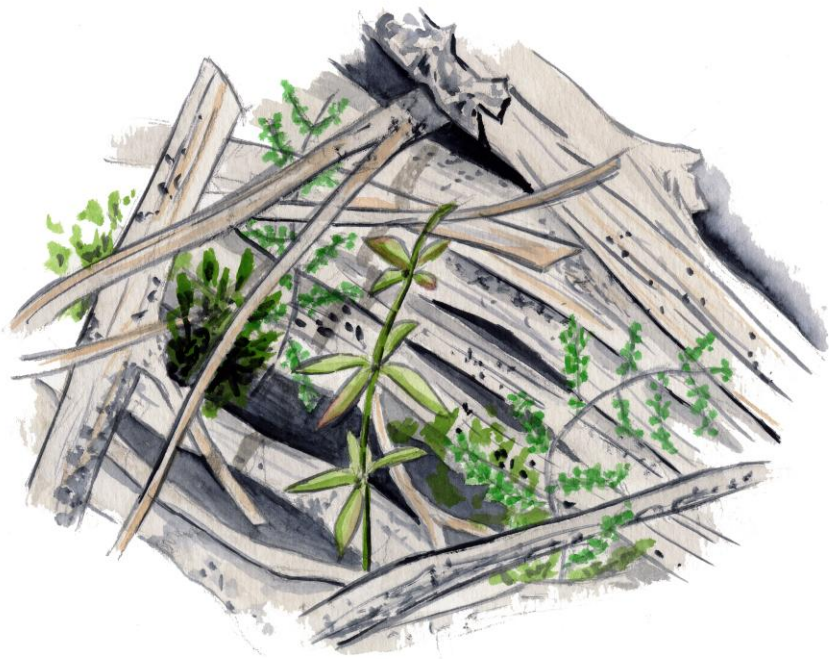
The importance of piling wood debris on the distribution of bird-dispersed plants in burned and logged Mediterranean pine forests

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Abstract

Plant dispersal and colonization are important processes in vegetation recovery after fire and, for several species, depend on the presence of frugivorous avian seed dispersers. In burned and salvage logged Mediterranean pine forests, the occurrence of disperser birds is enhanced by the presence of wood debris piles built as erosion barriers, which become seed dispersal foci. We hypothesized that the distribution of bird-dispersed plants after fire could be determined by the presence of wood piles. We analyzed the differences in species richness and overall cover of bird-dispersed plants, as well as the cover of the most common species, within piles and outside of them. We also tested if micro-environmental conditions (slope orientation and pile size) may influence plant occurrence. We found more species and overall cover within piles, when located on southern and south-eastern slopes, than outside of them. Moreover, taller piles showed higher values of bird-dispersed plant cover and richness. Therefore, wood piles may provide the necessary conditions for bird-dispersed plants' recruitment in dry burned areas. To our knowledge, this is the first reported man-made structure that clearly benefits the recovery of bird-dispersed plants in burned areas, and suggests that piles may be a useful restoration measure when logging Mediterranean burned forests.

Keywords: fire, seedling recruitment, postfire management, restoration.

Introduction

Plant regeneration strategies after fire involve resprouting from undamaged subterraneous organs or epicormic shoots, germination of heat-resistant seed banks and colonization from outside the burned area (Verdú 2000, Keeley & Fotheringham 2000). Though the resilience of postfire vegetation is ensured in many ecosystems by resprouting and germination, fire may also affect other important community processes. Among them, dispersal and colonization can alter population dynamics and influence the characteristics of individuals from distant origins recruited into the community

(Loveless & Hamrick 1984, Rey & Alcántara 2000). Since plants affected by fire initially dedicate far more effort to vegetative growth than to reproductive success, in the first years after fire, the seed dispersal of species with short-lived, large seeds that are killed by fire depends basically on the contribution of seed disperser animals from unburned areas. These animals, mostly frugivorous birds and some carnivorous mammals, contribute to seed dispersal by eating the fruit and defecating the seed afterwards (Herrera 1989, Herrera 1995). Alternatively, seeds may be transported directly to other areas (e.g., by ants) (Aronne & Wilcock 1994, Bas *et al*. 2009).

The drastic habitat alteration induced by fire causes changes in the distribution of Mediterranean avian seed dispersers (e.g. Pons & Prodon 1996, Herrando *et al*. 2003, Jacquet & Prodon 2009). Furthermore, in the Mediterranean basin, after pine forests burn they are usually salvage logged, a measure intended to harvest the profitable timber and minimize the economic losses caused by fire. Such postfire management constitutes a second disturbance within a short time interval, especially intense in clearcuts where the previous burned forest is turned into a fine-grained mosaic of grassland and shrubland. This additional habitat change also affects frugivorous bird dispersers, and its impact depends on altitude and on the habitat preferences of each species. For instance, in montane areas of southern Spain, Castro *et al*. (2010) reported a strong negative effect of salvage logging on bird dispersers, consisting mainly of *Turdus* thrushes. In contrast, in lowland Mediterranean burned areas, the most abundant frugivorous dispersers are *Sylvia* warblers (Herrando *et al*. 2003, Jacquet & Prodon 2009), which prefer the shrubland areas in the open habitat created by salvage logging rather than unlogged areas (Izhaki & Adar, 1997). Nevertheless, the regenerated shrubland is suitable for lowland dispersers only after significant vertical regrowth has taken place. Therefore, changes in the abundance and distribution of dispersers in recently-burned environments may hinder the seed dispersal of fleshy-fruited plants, whose regeneration in burned habitats is restricted to resprouting or to germination from a seed bank, with consequent limits on the recruitment of individuals from distant areas.

However, the management of wood debris that is not commercially profitable (branches and the thinnest pine logs) appears to be a key issue in terms of the persistence of avian seed dispersers in burned and logged pine forests (Rost *et al*. 2010). Whenever wood debris is not completely removed from the burned area (e.g. to produce biomass fuel), it may be left scattered on the ground, or in some cases heaped up into piles, which are sometimes built into log or branch erosion barriers, aimed at helping the restoration process (Robichaud *et al*. 2000). The construction of such barriers, designed to prevent erosion by runoff, is a relatively rare practice in burned and logged pine forests in the Mediterranean basin, and their effectiveness in reducing runoff is variable, depending on factors such as structural design, the combination with other mitigation treatments, the characteristics of the terrain and rainfall intensity (Marquès & Mora 1998, deWolfe *et al*. 2008, Robichaud *et al*. 2008). However, piles made up of logs and branches are a key microhabitat for frugivorous birds, which use them for foraging, perching and

shelter even quite soon after fire, when vegetation regrowth is just beginning (Rost *et al.* 2009).

The frequent use of wood piles by frugivorous birds may be behind the high density of dispersed seeds found beneath piles in burned and logged forests, which is equivalent to that found in unburned forests (Rost *et al.* 2009). The role of wood debris piles as seed dispersal foci suggests important consequences for the distribution of dispersed species in burned areas. If a significant part of dispersed seeds can be successfully established, this microhabitat-dependent seed rain may result in more individuals of bird-dispersed species being found in wood piles than in other microhabitats, which may mean a future heterogeneous plant distribution in areas where wood piles have been created. On a broader scale, this would indicate that burned and logged areas where such piles have been built up may hold more diverse plant communities than areas lacking these structures. Moreover, taking into account that dispersed plants produce fleshy fruits that are consumed (and dispersed) by several animal species, the effect of building wood piles may have implications at the ecosystem level. However, the establishment of dispersed seeds under piles will depend on various post-dispersal factors, such as seed predation rates, germination and emergence success, and seedling survival, all of which are shaped by many different habitat and microhabitat conditions (Kollmann *et al.* 2000).

The aims of this study were to test the following hypotheses: 1) bird-dispersed plant species will be more abundant within wood piles than outside of them and 2) extrinsic micro-environmental conditions (i.e. the orientation of the terrain slope) and intrinsic ones (such as the size of the piles) will have an influence in the effect that wood piles have on the establishment of dispersed plants and their distribution in burned and logged Mediterranean pine forests.

Methods

Study area

The study was conducted in Catalonia (NE Iberia), in three burned Aleppo pine (*Pinus halepensis*) forest sites: Montgrí (henceforth MO; 42°3'34" N, 3°7'22"E), Sant Llorenç (SL; 41°40'55" N, 02°1'53"E) and Cardona (CA; 41°54'31" N, 1°43'17" E). Since building wood debris piles into erosion barriers is, at present, uncommon in the study region, we could only find cases of this practice in these three recently-burned sites. All three sites have a Mediterranean climate (mean annual rainfall c. 600 mm, mean annual temperature 13-15ºC, with MO being slightly warmer than SL and CA). The substrate in MO and CA consists of limestone, while in SL it is sedimentary. CA burned in 2005 (1,438 ha), MO in 2004 (627 ha) and SL in 2003 (4,559 ha). All were severe canopy wildfires that killed the majority of pines and, after the fire, the majority of the snags were salvage logged. In certain areas of each site (20 ha in MO, 6 ha in SL and 5 ha in

CA), wood debris piles were built into log erosion barriers. The density of piles in these areas was high, reaching 100 piles/ha (Figure 1). Piles were set up at about 10 m apart between each other, had an average length of 7.4 m (±0.4 SE), an average height of 0.5 m (±1.3 SE) and an approximate width of 1 m.

Figure 1. Photograph showing an area where wood piles were built (courtesy of Alex Rollán and Joan Real, Equip de Biologia de la Conservació-Universitat de Barcelona).

Sampling design and field methods

We surveyed two areas in the MO burned area, one on a northern slope (8 ha), the other on a southern slope (8.5 ha) and two more areas in CA (1.5 ha) and SL (1.6 ha), both on south-eastern slopes. Unfortunately, in choosing these areas we were limited by the availability of areas where piles had been built, which did not occur in the same slopes in all sites. This fact made impossible to replicate adequately these conditions and therefore, to separate the effect of slope and site, an issue that we have to consider when interpreting the results. In each area, we sampled fleshy-fruited plant species in 100 sampling stations, 50 of them within piles (in-pile) and the other 50 outside of piles (out-pile). We used a UTM coordinate grid drawn on an aerial photograph of each pile area to locate the sampling stations, which were situated in the intersection of the grid lines. In MO, the grid was 50x50 m in size, while in CA and SL it was 25x25 m, due to the more reduced extent of the pile area. After finding each grid intersection (by GPS), the pile-out sampling station was placed in the exact intersection coordinates and the in-pile station was set in the nearest pile, with the condition that it was at least at 5 m from the out-pile point. When the intersection coordinates coincided with a pile, this point

became an in-pile station, while the out-pile station was set 5 m north from the pile, an orientation chosen with the purpose of standardizing the procedure. The 5 m distance was intended to reduce any possible influence between in-pile and out-pile paired stations, and was chosen considering that piles were set up with an approximate separation of 10 m to each other, thus being 5 m the maximum possible distance between an out-pile station and its paired in-pile one. We sampled fleshy-fruited plant cover at each station with a 1x1 m square delimiting a grid of 25 10x10 cm squares. The 1x1 m square was placed four times per station (two to the left and two to the right, starting from the station center and adjacent to each other), which made for a total of 100 squares per station. For each sampling station, we recorded the presence of fleshy-fruited plants in every 10x10 cm square, and then calculated the frequency of each bird-dispersed plant species cover in each station as the number of squares with presence of the species on the 100 squares. Furthermore, we measured the height of each pile in order to assess if pile height had an effect on fleshy-fruited plant cover. At each in-pile station, we measured pile height at four points, each one coinciding with the center of the four cover measurements (i.e. each point separated by 1 m from the next), and obtained the average height of each pile, which we used for further analyses. Sampling was carried out in the spring of 2009, between four and six years after fire, depending on the site.

Data analysis

We analyzed the effect of piles and pile height on species richness of fleshy fruited plants and total cover, as well as on the cover of the most common species (those occurring in more than 10% of the stations of a given pile area; N>10). When a species was absent from one site we excluded this site from the analysis. When analyzing two or three sites, we used generalized linear mixed models (GLMM), with site included as a random factor in order to control for spatial-associated variability. When only one site was analyzed, generalized linear models (GLM) were performed. Pile (in-pile/out-pile) was a fixed- effects factor in all analyses. Slope (N, S or SE) was also included as a fixed-effects factor if the analyzed data included more than one slope category. In that case, pile was included in the analysis nested within slope. We used the same analysis structure for pile height, but excluded all out-pile stations. In both analyses, we selected a Poisson error distribution —or quasipoisson when the first was overdispersed (Crawley, 2000)— and a log link. For the GLMM, we used the PQL approximation. All analyses were performed with the R 2.9.2 (R Development Core Team 2009) program. We took $p < 0.05$ as being statistically significant in all cases.

Table 1. Percentage of occurrence of fleshy fruited plants in the stations, calculated from 50 pile-in (in) and 50 pile-out (out) stations per site. Site's slope shown in parenthesis.

Scientific name	Common name	MO (N) in	MO (N) out	MO (S) in	MO (S) out	SL (SE) In	SL (SE) out	CA (SE) in	CA (SE) out
Rubia peregrina	Wild madder	74	56	84	46	72	40	78	34
Asparagus acutifolius	Wild asparagus	60	62	48	36	12	4	0	0
Smilax aspera	Rough bindweed	56	54	22	12	10	6	0	0
Lonicera implexa	Evergreen honeysuckle	32	42	6	2	8	4	0	0
Rubus ulmifolius	Bramble	6	4	0	0	28	20	20	2
Pistacia lentiscus	Lentisk	6	0	0	0	14	10	16	8
Rhamnus alaternus	Evergreen buckthorn	4	4	4	0	10	4	0	0
Ligustrum vulgare	Wild privet	0	0	0	0	10	10	0	0
Daphne gnidium	Spurge-flax	0	0	0	0	8	8	0	0
Arbutus unedo	Strawberry tree	0	0	0	0	0	0	4	10
Crataegus monogyna	Hawthorn	8	4	0	0	0	0	0	0
Ruscus aculeatus	Butcher's broom	0	8	0	0	2	0	0	0
Phyllirea latifolia	Mock privet	2	6	0	0	0	0	0	0
Juniperus oxycedrus	Prickly juniper	0	0	0	0	0	2	2	2
Olea europaea	Olive tree	2	4	0	0	0	0	0	0
Rosa sempervirens	Evergreen rose	0	4	0	0	2	0	0	0
Viburnum tinus	Laurestine	4	2	0	0	0	0	0	0
Pistacia terebinthus	Terebinth tree	0	2	0	0	0	0	0	0
Prunus spinosa	Blackthorn	0	0	0	0	0	0	2	0

Results

We found 19 plant species with fleshy fruit, potentially dispersed by birds (Table 1). The six most common species were *Rubia peregrina*, *Asparagus acutifolius*, *Smilax aspera*, *Lonicera implexa*, *Rubus ulmifolius* and *Pistacia lentiscus*, and their cover was analyzed individually.

The GLMM and GLM results showed a positive and significant effect of piles on the species richness of fleshy-fruited plants, on all fleshy-fruited plant cover and on *Rubia peregrina*, *Asparagus acutifolius* and *Rubus ulmifolius* cover (Table 2). We observed that slope orientation had a strong influence on the effect that the wood piles had on species richness, total fleshy-fruited plant cover and *A. acutifolius* cover. In these cases, southern and south-eastern slopes showed higher in-pile than out-pile values, whereas in stations located on northern slopes, we did not observe such differences (Figure 2). However, we did not find any influence of slope in the case of *R. peregrina* cover.

Furthermore, we found a positive effect of pile height on richness, total cover and *A. acutifolius* cover and this also depended on slope orientation, since this positive effect was only observed in southern and south-eastern slopes and not in northern ones (Figure 3).

Discussion

We found that the distribution of some bird-dispersed plants in burned and logged pine forests is highly influenced by new structures created by postfire management. Our results show that the presence of wood debris piles is favorable to the abundance of, at least, *Rubia peregrina*, *Asparagus acutifolius* and *Rubus ulmifolius* —species which are likely to be far less represented in the absence of piles, especially in southern and south-eastern slopes. This is a remarkable result, because it means that a man-made structure can help the recovery of several Mediterranean plant species after fire and postfire management. Although it was not possible to know if the individuals established beneath piles were actually regenerated from a bird-dispersed seed or by resprouting, we found that —whatever the regeneration strategy used— piles favored a greater bird-dispersed plant cover.

The success of the plant recruitment process in its different stages (seed dispersal, seed survival, germination, seedling survival and establishment) depends on several factors that are determined by habitat and microhabitat quality, e.g. the presence of seed or seedling predators, soil characteristics, distance from parent plant or presence of perches (Kollmann 2000, Gómez-Aparicio 2008). Although being aware of the lack of independence in our data, which does not effectively allow to separate the effects of slope and site, the fact that the positive effect of wood piles was found only in southern and south-eastern slopes but not in the northern slope area, suggests that the orientation can influence the effect of piles on plant cover. In areas oriented to the south, irradiance is higher and soil is usually thinner, with higher temperature and lower humidity than in northern slopes, where vegetation regenerates faster after fire (Raftoyannis & Spanos 2005). Southern slope soils are probably less suitable for seed germination and seedling emergence because water availability in the soil is critical in this process (Evans & Etherington 1990). However, on these drier slopes, soil under wood piles may present different micro-environmental conditions than those of the surrounding ground in the short time after fire, such as, for example, much lower irradiance. Therefore, the soil would be more humid and less compact, as is the case beneath nurse shrubs (Verdú & García-Fayos 1996, García-Fayos & Verdú 1998, Gómez-Aparicio 2008), creating a favorable situation for seed germination and seedling establishment. The fact that the cover of bird-dispersed plants was almost as high in piles situated in southern slopes as it is in northern slopes (either in piles or outside them) would support this idea. If so, such effect could be independent of the plant dispersal strategy, being likely to be observed in other species as well, which could be worth to be investigated in the future. The fact that taller piles have a greater effect on fleshy-fruited species richness and cover (Figure 3) may also be related to this effect, since taller piles may maintain better micro-environmental conditions for seed germination than shorter piles. On the other hand, on northern slopes, *R. peregrina* was the only species that was positively affected by slash piles. This may indicate that this species would not be as influenced by different micro-environmental conditions between slopes as other plant species. The larger cover of this species in piles may be more related to other positive effects of piles

Table 2. Effect of pile presence and pile height on seven response variables (richness, total cover, and six fleshy-fruited species), in each slope (N, S and SE). Significant results are highlighted in bold. We also present the model estimate in order to show the sign of each effect.

In/out of pile	N		S		SE	
	Estimate	*p*	*Estimate*	*p*	*Estimate*	*p*
Richness	0.007	0.944	0.535	**0.001**	0.579	**<0.001**
Total cover	0.135	0.479	1.245	**<0.001**	0.552	**0.001**
Rubia peregrina	1.163	**<0.01**	1.826	**<0.001**	1.313	**<0.001**
Asparagus acutifolius	0.216	0.445	0.683	**0.040**	-	-
Smilax aspera	-0.105	0.765	0.971	0.082	-	-
Lonicera implexa	-0.719	0.067	-	-	-	-
Rubus ulmifolius	-	-	-	-	0.990	**0.039**
Pistacia lentiscus	-	-	-	-	-0.402	0.458

Pile height	*Estimate*	*p*	*Estimate*	*p*	*Estimate*	*p*
Richness	-0.003	0.531	0.021	**0.018**	-0.005	0.302
Total cover	-0.001	0.370	0.023	**0.041**	-0.001	0.930
Rubia peregrina	-0.018	0.117	0.007	0.591	0.002	0.713
Asparagus acutifolius	0.005	0.639	0.060	**0.001**	-	-
Smilax aspera	0.000	0.986	0.029	0.306	-	-
Lonicera implexa	-0.032	0.064	-	-	-	**-**
Rubus ulmifolius	-	-	-	-	0.004	0.742
Pistacia lentiscus	-	-	-	-	0.023	0.251
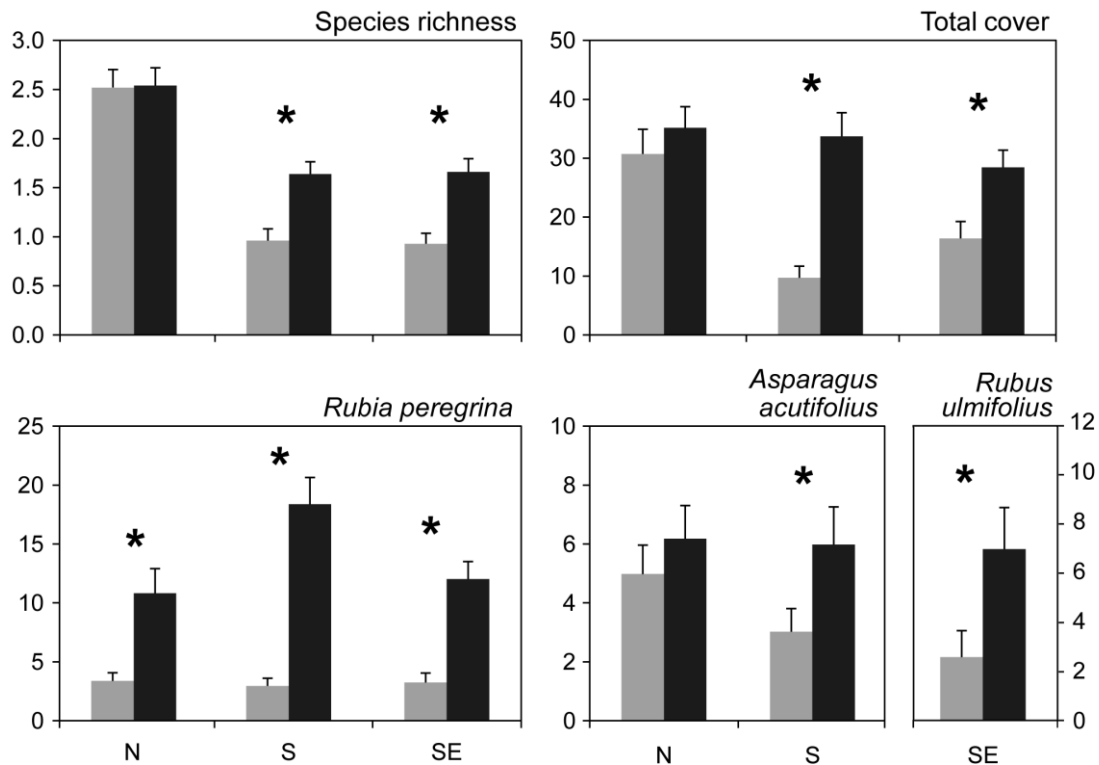
Figure 2. Mean richness, total cover and covers of *Rubia peregrina*, *Asparagus acutifolius* and *Rubus ulmifolius* outside of piles (grey bars) or within piles (black bars), on each slope (N: north, S: south and SE: south-east). Error bars show standard error and significant effects are shown with an asterisk.

on species recruitment, e.g. a greater seed rain in these structures, the fact that it is a climbing plant and the piles provide structural support for its growth, reduced competition with heliophilous shrubland species beneath these piles, or some possible role of piles in preventing seed and seedling predation.

In fact, wood piles may have an important influence on post-dispersal seed predation and seedling predation in recently burned areas. The closed structure of these piles may offer protection to seeds against granivorous birds, which can be abundant after fire (Thibault & Prodon 2006), and ungulates, as has been seen in the case of wild boar (*Sus scrofa*), which cannot easily reach the diaspores if they are situated under a layer of wood debris (Puerta-Piñero *et al*. 2010). However, in burned Mediterranean forests, the predation pressure exerted by these groups on seeds appears to be less than that of ants and rodents (Ordóñez & Retana 2004, Puerta-Piñero *et al*. 2010) and wood piles do not offer much protection against ants and rodents. Predation by rodents is usually higher in protected microhabitats than it is in open sites while the contrary happens with granivorous or harvester ants (Hulme 1997), although in early successional habitats, granivorous ants do not show clear microhabitat preferences (Rey *et al*. 2002). This could mean that both rodents and ants may be important seed predators under piles. Moreover, wood piles may represent important shelters for rabbits (Rollán & Real 2010) and therefore there may be significant seedling predation by these herbivores beneath the piles. However, recruitment is determined by seed and seedling predation only in seed-limited areas, while predation has a minimal effect in safe sites where recruitment is not seed-limited (Andersen 1989, Calviño-Cancela 2007). The high density of dispersed seeds found below wood piles compared to other microhabitats of burned and unburned Mediterranean pine forests (Rost *et al*. 2009) would support the idea that the piles do not show seed limitation. Therefore, if we can consider them as safe sites, predation would not be a determinant constraint for seedling recruitment.

On the other hand, if wood debris piles were useful for reducing soil erosion by runoff, richer soils would be found under piles than elsewhere, which would also favor plant establishment. Moreover, whatever the effect against erosion, the attraction of birds to branch piles, as perches or as a foraging microhabitat (Rost *et al*. 2010), would mean that the soil under the piles will also be richer because of fertilization by bird droppings.

In turn, a higher abundance of fleshy-fruited plants in wood piles implies that, in the future, such structures may represent important food resources for fruit-eating fauna (and also other species that depend on such fauna) —food that is otherwise scarce, especially in the driest areas. If frugivorous disperser birds have an important role in determining the presence of these plants in burned areas (Rost *et al*. 2009), the same bird species responsible for the establishment of certain fleshy-fruited plants will benefit in the future from the process they started some years before.

In conclusion, tall piles of wood debris built in burned areas, intended to prevent soil erosion, may have other unplanned restoration results, such as facilitating the regeneration of fleshy-fruited plant species. This process was found to be especially
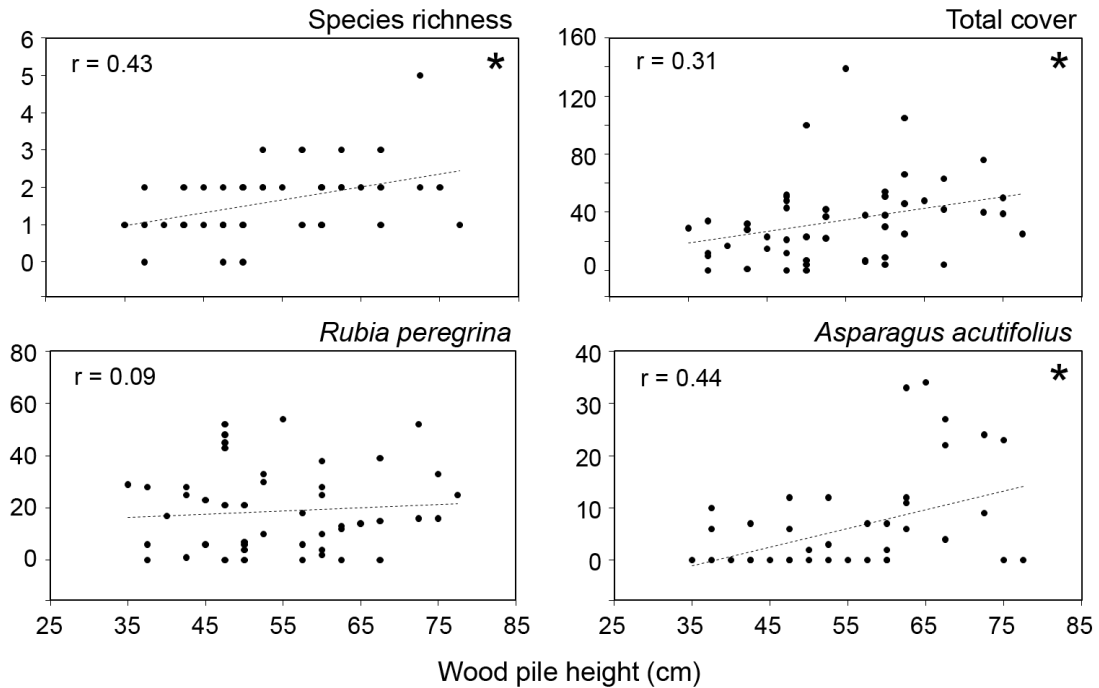
Figure 3. Effect of wood pile height on richness, total cover and covers of *Rubia peregrina* and *Asparagus acutifolius*, on southern slopes. Significant relationships (p<0.05) are shown with an asterisk.

important on southern slopes —with more stressful conditions regarding water availability— where the presence of a favorable micro-environment under piles could determine the abundance of such plants, an interesting point that needs to be confirmed by further experimental research. Therefore, piles can represent safe sites for fleshy-fruited, bird-dispersed plant germination, seedling survival and establishment in burned areas, as well as being important seed dispersal foci in the same environments. It also brings potential benefits for fruit-eating fauna in burned areas. Therefore, we highly recommend including the creation of wood piles in the management and restoration strategies of burned and logged Mediterranean pine forests.

Acknowledgements

References

Andersen, A.N. 1989. How important is seed predation to recruitment in stable population of long-lived perennials? Oecologia 81, 310-315.

Aronne, C. & Wilcock, C.C. 1994. First evidence of myrmecochory in fleshyfruited shrubs of the Mediterranean region. New Phytologist 127, 781-88.

Bas, J.M., Oliveras, J. & Gómez, C. (2009) Myrmecochory and short-term seed fate in Rhamnus alaternus: Ant species and seed characteristics. Acta Oecologica 35, 380-384.

Calviño-Cancela, M. 2007. Seed and microsite limitations of recruitment and the impacts of post-dispersal seed predation at the within population level. Plant Ecology 192, 35-44.

Castro, J., Moreno-Rueda, G. & Hódar, J.A. 2010. Experimental test of postfire management in pine forests: impact of salvage logging versus partial cutting and nonintervention on bird–species assemblages. Conservation Biology 24, 810-819.

Crawley, M.J. 2000. Statistical computing: an introduction to data analysis using S-Plus. John Wiley & Sons Ltd., Chichester.

deWolfe, V.G., Santi, P.M., Ey, J. & Gartner, J.E. 2008. Effective mitigation of debris flows at Lemon Dam, La Plata County, Colorado. Geomorphology 96, 366-377.

Evans, J.R. & Etherington, C.E. 1990. The effect of soil water potential on seed germination of some British plants. New Phytologist 115, 539-548.

García-Fayos, P. & Verdú, M. 1998. Soil seed bank, factors controlling germination and establishment of a Mediterranean shrub: Pistacia lentiscus L. Acta Oecologica 19, 357-366.

Gómez-Aparicio, L. 2008. Spatial patterns of recruitment in Mediterranean plant species: linking the fate of seeds, seedlings and saplings in heterogeneous landscapes at different scales. Journal of Ecology 96, 1128-1140.

Herrando, S., Brotons, L. & Llacuna, S. 2003. Does fire increase the spatial heterogeneity of bird communities in Mediterranean landscapes? Ibis 145, 307-317.

Herrera, C.M. 1989. Frugivory and seed dispersal by carnivorous mammals, and associated fruit characteristics, in undisturbed Mediterranean habitats. Oikos 55, 250-62.

Herrera, C.M. 1995. Plant–vertebrate seed dispersal systems in the Mediterranean: ecological, evolutionary, and historical determinants. Annual Review of Ecology, Evolution and Systematics 26, 705-727.

Hulme, P.E. 1997. Post-dispersal seed predation and the establishment of vertebrate dispersed plants in Mediterranean scrublands. Oecologia 111, 91-98.

Izhaki, I. & Adar, M. 1997. The effects of post-fire management on bird community succession. International Journal of Wildland Fire 7, 335-342.

Jacquet, K. & Prodon, R. 2009. Measuring the postfire resilience of a bird–vegetation system: a 28-year study in a Mediterranean oak woodland. Oecologia 161, 801-811.

Keeley, J.E. & Fotheringham, C.J. 2000. Role of fire in regeneration from seed. In: Fenner, M. (ed.). Seeds: the ecology of regeneration in plant communities. C.A.B. International, Wallingford.

Kollmann, J. 2000. Dispersal of fleshy-fruited species: a matter of spatial scale? Perspectives in Plant Ecology, Evolution and Systematics 3, 29-51.

Loveless, M. & Hamrick, J. 1984. Ecological determinants of genetic structure in plant populations. Annual Review in Ecology Evolution and Systematics 15, 65–95.

Marquès, M.A. & Mora, E. 1998. Effects on erosion of two post-fire management practices: clear-cutting versus non-intervention. Soil & Tillage Research 45, 433-439.

Ordóñez, J.L. & Retana, J. 2004. Early reduction of post-fire recruitment of Pinus nigra by post-dispersal seed predation in different time-since-fire habitats. Ecography 27, 449-458.

Pons, P. & Prodon, R. 1996. Short term temporal patterns in a Mediterranean shrubland bird community after wildfire. Acta Oecologica 17, 29-41.

Puerta-Piñero, C., Sánchez-Miranda, A., Leverkus, A. & Castro, J. 2010. Management of burnt wood after fire affects post-dispersal acorn predation. Forest Ecology and Management, 260: 345-352.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Raftoyannis, Y. & Spanos, I. 2005. Evaluation of log and branch barriers as post-fire rehabilitation treatments in a Mediterranean pine forest in Greece. International Journal of Wildland Fire 14, 183-188.

Rey, P.J. & Alcántara, J.M. 2000. Recruitment dynamics of a fleshy-fruited plant (*Olea europaea*): connecting patterns of seed dispersal to seedling establishment. Journal of Ecology 88, 622-633.

Rey, P.J., Garrido, J.L., Alcántara, J.M., Ramírez, J.M., Aguilera, A., García, L., Manzaneda, A.J. & Fernández, R. 2002. Spatial variation in ant and rodent post-dispersal predation of vertebrate-dispersed seeds. Functional Ecology 16, 773-781.

Robichaud, P.R., Beyers, J.L. & Neary, D.G. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. USDA Forest Service, Fort Collins.

Robichaud, P.R., Wagenbrenner, J.W., Brown, R.E., Wohlgemuth, P.M. & Beyers, J.L. 2008. Evaluating the effectiveness of contour-felled log erosion barriers as a post-fire runoff and erosion mitigation treatment in the western United States. International Journal of Wildland Fire 17, 255–273.

Rollán, A. & Real, J. 2010. Effect of wildfires and post-fire forest treatments on rabbit abundance. European Journal of Wildlife Research, in press. DOI : 10.1007/s10344-010-0412-y.

Rost, J., Pons, P. & Bas, J.M. 2009. Can salvage logging affect seed dispersal by birds into burned forests? Acta Oecologica 35, 763-768.

Rost, J., Clavero, M., Bas, J.M. & Pons, P. 2010. Building wood debris piles benefits avian seed dispersers in burned and logged Mediterranean pine forests. Forest Ecology and Management 260, 79-86.

Thibault, J-C. & Prodon, R. 2006. Seed predation by birds shortly after a wildfire in a Corsican pine forest. International Journal of Wildland Fire 15, 81-86.

Verdú, M. 2000. Ecological and evolutionary differences between Mediterranean seeders and resprouters. Journal of Vegetation Science 11, 265-268.

Verdú, M. & García-Fayos, P. 1996. Nucleation processes in a Mediterranean bird-dispersed plant. Functional Ecology 10, 275-280.

Discussió general i conclusions

DISCUSSIÓ GENERAL I CONCLUSIONS

El conjunt de treballs presentats en aquesta tesi estudien com és l'efecte de dues pertorbacions severes, el foc i la tala dels arbres després de l'incendi, sobre l'avifauna i sobre alguns processos ecològics que en depenen. Els resultats obtinguts posen de manifest la importància que té la gestió forestal post-incendi, pel gran canvi que es produeix en l'hàbitat que es tradueix en un canvi molt significatiu en la comunitat d'ocells que ocupa aquestes àrees i que afecta processos a nivell d'ecosistema on els ocells estan implicats. Les estructures resultants de la gestió (com els pilons de branques) influeixen també en l'ús que fan els ocells de l'hàbitat, la qual cosa repercuteix en processos on aquests interactuen amb altres espècies, com la dispersió de llavors d'algunes plantes. Per tant, els resultats recollits en aquesta tesi són rellevants atès que aporten informació nova i aplicada, que pot ser utilitzada per millorar els criteris actuals de gestió post-incendi dels boscos mediterranis. A continuació es discuteixen i es relacionen els resultats dels diferents capítols d'aquesta tesi, i es proposen recomanacions concretes de gestió.

La resposta de l'avifauna a dues pertorbacions severes

La resposta de la fauna a les pertorbacions ecològiques depèn bàsicament de la magnitud o severitat d'aquestes (Karr & Freemark 1985). Les respostes poden anar des de petites modificacions fisiològiques a nivell individual fins a canvis a nivell de població (densitat, reproducció, distribució espacial, probabilitat d'extinció,...), de comunitat (composició d'espècies, estructura tròfica,..) i fins i tot a nivell d'ecosistema, si la resposta de les espècies afecta processos en els que aquestes estan implicades (Karr & Freemark 1985). En el cas de l'avifauna, l'efecte d'una pertorbació com el foc en ecosistemes forestals ja és força conegut, i implica canvis importants tan a nivell poblacional com de comunitat (Hutto 1995, Herrando *et al.* 2003, Jacquet & Prodon 2009); uns canvis que estan modulats per la severitat del foc (Barlow & Peres 2004, Smucker *et al.* 2005).

Si al foc s'hi suma una segona pertorbació de la magnitud d'una tala post-incendi, la transformació de l'hàbitat és encara més dràstica i produeix canvis encara més pronunciats a nivell poblacional i de comunitat, com mostren els resultats dels capítols 1 i 3 d'aquesta tesi i com han provat estudis anteriors (Morissette *et al.* 2002, Izhaki & Adar 1997, Castro *et al.* 2010). A nivell poblacional, la presència de moltes espècies (de fet, la majoria d'elles) depèn de la tala post-incendi, algunes de les quals colonitzen el nou hàbitat o s'extingeixen localment després de la intervenció. A nivell de comunitat, aquestes respostes poblacionals es visualitzen en un canvi important i ràpid en la composició d'espècies.

Per altra banda, l'estudi d'ambdues pertorbacions dut a terme al capítol 1 permet veure com és el seu efecte, tant per separat com de manera combinada, sobre les espècies i

sobre la comunitat d'ocells. Tot i l'important efecte de la tala en la presència de la majoria d'espècies, i per tant en la composició de la comunitat, altres paràmetres com la riquesa i l'abundància es veuen més afectats pel pas del temps des del foc. Aquesta estabilitat en riquesa i densitat després de la tala ja havia estat observada per Izhaki & Adar (1997) a l'est del Mediterrani, també en pinedes de pi blanc de terra baixa (a uns 300 m s.n.m.). Tanmateix, en boscos de pi situats a més altitud (1.500-2.000 m), Castro *et al.* (2010) van observar que tant la riquesa com l'abundància d'ocells disminuïen després de la tala, cosa que podria estar relacionada amb una major lentitud de colonització d'ambients cremats a muntanya per part dels ocells d'espais oberts, on arriben a les seves màximes abundàncies més tard que a terra baixa, entre una i dues dècades després del foc (Pons & Clavero 2010, dades del Pirineu).

D'altra banda, l'efecte d'aquestes dues pertorbacions és diferent segons el grup d'espècies estudiat. Separant aquestes en forestals i d'espais oberts-agrícoles, es posa de manifest com les primeres disminueixen tant si es tallen els arbres com si no es fa. Per tant, el canvi d'hàbitat causat pel foc, en si mateix ja implica que la densitat d'aquestes espècies disminueixi des del primer any després del foc. Per contra, si bé l'obertura de l'hàbitat causada pel foc també beneficia les espècies d'espais oberts (presents també en zones no talades), aquestes es veuen més clarament beneficiades per la tala dels arbres, augmentant especialment en densitat durant el segon i tercer any en aquells llocs que s'han talat. En global, aquest increment és de prop del 70%, i possiblement es degui a la combinació de la continuació de la colonització i de la reproducció local de la població que havia colonitzat l'àrea durant el primer i segon any després del foc, anys entre els quals la seva densitat varia molt menys. Aquest fet probablement sigui la causa que també els valors de riquesa i densitat millorin entre la segona i la tercera primavera, respecte del pas entre la primera i la segona; un fet especialment obvi en els indrets on els arbres cremats es van tallar.

Per altra banda, seria d'esperar que els canvis en l'avifauna que tenen lloc a nivell de població i comunitat es traduïssin en canvis en aquells processos de l'ecosistema on participen els ocells. La dispersió de llavors és un d'aquests processos, en el que els ocells frugívors tenen un paper important en ecosistemes mediterranis (Herrera 1984). Tanmateix, aquest tipus de mutualismes planta-animal es poden veure interromputs quan una pertorbació afecta l'espècie dispersora (e.g. mamífers, cacera i fragmentació: Peterson *et al.* 1998, Wright & Duber 2001; rèptils i introducció d'espècies exòtiques: Traveset & Riera 2005). Tanmateix fins el moment no hi havia informació sobre com influeixen dues pertorbacions com el foc i la gestió forestal posterior sobre la dispersió de llavors per ocells, tot i que és un tema ja plantejat per alguns autors (e.g. Castro *et al.* 2010). Com mostren els resultats dels capítols 3, 4 i 5 d'aquesta tesi, la transformació de l'hàbitat i la disponibilitat o no de certes estructures com a resultat tan del foc com de la gestió posterior tenen un efecte important sobre aquest procés. Tanmateix, és interessant (i útil de cara a la gestió) el fet que siguin estructures com els pilons de branques, que podríem qualificar d'artificials, les que afavoreixin la dispersió de llavors per ocells frugívors, i no tant els arbres que es deixen de tallar, que sovint han estat considerats

elements importants per a la fauna (Hutto 2006). Aquests resultats i les seves implicacions es comenten amb més detall més endavant.

Canvis d'espècies i implicacions de conservació

Com es descriu als capítols 1 i 3, el canvi dràstic en l'estructura de l'hàbitat que es produeix com a resultat de la tala post-incendi és el factor que determina de manera més important la composició de la comunitat d'ocells a curt terme, en boscos de pi blanc cremats. La presència de més de la meitat de les espècies més freqüents d'aquestes zones es veu condicionada per la tala o no dels arbres cremats, tant a l'època reproductora com a l'hivern. D'aquestes espècies, les que es veuen afectades de manera més negativa són les de requeriments més forestals, les quals necessiten la presència d'arbres cremats per tal de mantenir-se després de l'incendi (Llimona *et al.* 1993, Izhaki & Adar 1997, Kotliar *et al.* 2002, Morisette *et al.* 2002). En aquest sentit, com es mostra al capítol 2, les espècies més perjudicades per la tala post-incendi són, concretament, les que més depenen dels arbres cremats com a lloc potencial de nidificació, d'alimentació o com a component estructural del seu hàbitat, en un patró que es manté entre regions biogeogràfiques diferents, com la Conca Mediterrània i les Muntanyes Rocalloses nord-americanes.

No obstant, també hi ha una bona quantitat d'ocells que es beneficien de la tala dels pins cremats. Es tracta d'espècies pròpies de zones agrícoles i altres ambients oberts amb relativament poca cobertura de vegetació, característiques que es troben en les zones cremades durant els primers anys després del foc. Aquestes espècies tenen capacitat per colonitzar àrees cremades (Herrando *et al.* 2002, Pons & Bas 2005, Brotons *et al.* 2008) en el cas de tenir poblacions relativament properes a l'incendi (Brotons *et al.* 2005). Com mostren els resultats del capítol 1, la tala dels arbres potencia la presència d'aquest grup d'ocells d'espais oberts. Aquestes espècies colonitzen el nou hàbitat molt ràpidament, trobant-s'hi ja durant la primavera següent a la tala. En qualsevol cas, la tala post-incendi causa una substitució clara d'espècies, malgrat que no s'observi una davallada en el nombre net d'espècies ni en la densitat d'ocells, en el global dels tres anys després del foc.

D'altra banda, en el capítol 1 també es descriu com aquest canvi de la composició d'espècies fa canviar també l'índex de conservació de la comunitat. L'eliminació dels arbres cremats té com a conseqüència un augment en el valor de conservació de la comunitat d'ocells, beneficiant un conjunt d'espècies d'espais oberts i agrícoles que presenten un estatus de conservació força desfavorable a escala europea (e.g. el trobat *Anthus campestris*, el cruixidell *Miliaria calandra*, el cotoliu *Lullula arborea* o el passerell *Carduelis cannabina*; BirdLife International 2004). Les espècies d'ambients agrícoles, especialment, han experimentat una davallada molt important a gairebé tot Europa des dels anys 80, que oscil·la entre un 53% i un 25% de la seva població (PECBMS 2009). Les causes d'aquest declivi s'atribueixen, per una part, a canvis en la

Figura 1. El cotoliu *Lullula arborea* és un ocell típic d'espais oberts, que es veu afavorit per la tala dels pins cremats. A més a més, aquesta espècie té un estatus de conservació desfavorable a Europa.

gestió de l'agricultura, que s'ha intensificat i especialitzat. Això ha comportat una tendència a la homogenització de conreus, la pèrdua de marges, de guarets i de prats, i un increment en la utilització de pesticides i fertilitzants (Donald *et al.* 2001). D'altra banda, els canvis en els usos del sòl també han jugat un paper en la pèrdua d'hàbitat per a aquestes espècies. L'abandonament de les activitats agràries i el despoblament de zones rurals, especialment en els relleus i les àrees més improductives de tot Europa, ha portat a que zones antigament cultivades s'hagin convertit en boscos i matollars. Amb aquesta transformació, els ocells lligats a espais oberts han perdut part del seu antic hàbitat (Preiss *et al.* 1997, Sirami *et al.* 2007). Amb criteris de conservació europeus, per tant, es pot dir que la tala post-incendi pot ser una gestió de l'hàbitat positiva per a la conservació dels ocells d'ambients oberts, almenys en zones de mosaic agro-forestal com les estudiades, on les poblacions font d'espècies colonitzadores es troben a poca distància de la zona cremada. De tota manera, aquest efecte és relativament efímer, limitat només als primers anys després del foc. Després, la regeneració progressiva de la vegetació –que serà més o menys ràpida en funció de l'aridesa de la zona- porta a un tancament del matollar que fa que l'hàbitat deixi de ser tan favorable per a espècies d'espais oberts, beneficiant a tallarols i altres ocells propis de la brolla (Jacquet & Prodon 2009, Pons & Clavero 2010). D'aquestes espècies de matollar, per cert, la tallareta cuallarga *Sylvia undata* és l'ocell que té un estatus de conservació més preocupant d'entre totes les espècies trobades en aquesta tesi: es troba com a proper a l'amenaça a nivell global (BirdLife International 2008). Aquest ocell, tot i que no està afectat positivament per la tala post-incendi a la primavera, a l'hivern sí que prefereix zones cremades sense pins drets i amb pilons de branques, com es veu al capítol 3, de manera que la tala post-incendi no la perjudicaria, sinó més aviat el contrari.

Malgrat tot, també hi ha espècies amb un estatus de conservació desfavorable a Europa i que es veuen perjudicades per la tala post-incendi. Aquest és el cas de la tórtora *Streptopelia turtur*, molt afectada per la tala com es veu al capítol 1, de la mallerenga

emplomallada *Parus cristatus* i del papamosques gris *Muscicapa striata*, que estan relacionats amb la presència de peus cremats drets. En qualsevol cas, si les zones talades són un hàbitat poc favorable per a aquests ocells respecte de zones no talades, es pot afirmar el mateix en comparar pinedes cremades amb no cremades, on són molt més abundants (Herrando *et al.* 2003, Ukmar *et al.* 2007). De fet, la tórtora no es pot considerar un ocell estrictament forestal, sinó més aviat propi de zones "de mosaic" on s'intercalen boscos i altres tipus d'hàbitat, especialment zones obertes (Snow & Perrins 1998), de manera que una certa extensió d'aquestes àrees obertes resultat del foc i de la tala podria no ser-li tan negativa. Tampoc es pot descartar que la tórtora aprofiti els claps de pins no cremats que van quedar dins del perímetre dels incendis com a llocs de nidificació, i que utilitzi les zones més obertes per alimentar-se. Les mallerengues, al seu torn, van ser observades alimentant-se de les restes de pinyons que encara quedaven als pins cremats. Desconeixem, però, quin èxit poden tenir com a reproductores dins la zona cremada, on els recursos per a ocells que s'alimenten a les capçades, com aquest, poden ser escassos. D'altra banda, considerant una escala més àmplia que la del propi bosc cremat, i en un context de canvi d'usos del sòl on la superfície forestal va en augment, la presència d'aquestes espècies més forestals probablement no es vegi massa compromesa per la tala de boscos cremats. En qualsevol cas, seria molt interessant realitzar estudis més concrets i a diferents escales per veure com el foc i la tala afecten realment l'ús de l'hàbitat que fan aquestes i altres espècies d'ocells forestals.

Figura 2. La mallerenga emplomallada *Parus cristatus* és una espècie de requeriments més aviat forestals que està en una situació de conservació desfavorable a Europa, i que és present a les pinedes cremades.

Tanmateix, les conclusions relatives a l'efecte de la tala post-incendi sobre temes de conservació podrien haver estat unes altres d'haver treballat amb uns criteris diferents dels del SPEC europeu. A nivell català o espanyol, per exemple, la majoria d'ocells d'espais oberts no presenten tendències poblacionals negatives –alguns fins i tot augmenten- ni estan amenaçats a aquesta escala (Martí & del Moral 2003, Estrada *et al.* 2004, ICO 2010). Per què utilitzar el criteri SPEC i no criteris de llista vermella UICN a Catalunya o Espanya, per exemple? En primer lloc, utilitzar una escala regional

continental té l'avantatge d'evitar el considerar com a amenaçada una espècie que pot ser molt escassa a una escala local però que a una escala més gran no té problemes greus de conservació, i que per tant, no hauria de rebre esforços excessius per mantenir-la. Per altra banda, el sistema SPEC, té en compte la categorització de la llista vermella regional europea (IUCN 2003), l'amplia i adapta el grau d'informació disponible per a les poblacions europees, incloent, per exemple, tendències poblacionals més llargues o definint categories amb llindars de declivi més baixos. D'aquesta manera, espècies que no estan amenaçades segons els criteris de llista vermella regional, per exemple per haver tingut tendències poblacionals estables en els darrers deu anys o tres generacions (IUCN 2001), però que no s'han recuperat de declivis importants anteriors, apareixerien amb estatus desfavorable segons els criteris SPEC. Finalment, el sistema SPEC té la virtut de la simplicitat, classificant les espècies en quatre categories segons si el seu estatus és favorable o no, segons la responsabilitat d'Europa en la seva conservació, i segons si l'espècie està amenaçada a nivell global (BirdLife International 2004). Per tot això, un índex de conservació basat en els criteris SPEC pot ser més adequat que un de basat en altres sistemes de categorització o en àmbits geogràfics més restringits.

De totes maneres, és interessant posar les tendències dels ocells d'espais oberts, que ara són objectius de conservació, en un context temporal més ampli, fins i tot històric. La davallada d'ocells d'ambients agrícoles (i l'increment dels forestals) ha estat documentada a Europa des dels anys seixanta i setanta fins a l'actualitat (Vorisek *et al.* 2008, PECBMS 2009), i s'ha atribuït bàsicament a l'intensificació agrícola i als canvis en els usos del sòl. És a dir, es tracta d'espècies les poblacions de les quals varien d'acord amb els canvis en el paisatge i l'hàbitat, que tenen com a causa fonamental la manera en què l'home ha explotat els recursos naturals. Agafant períodes de temps diferents, les tendències dels ocells d'espais oberts haurien estat també diferents, i la seva situació de conservació també. Per exemple, des de l'edat mitjana fins als inicis del segle XX la superfície forestal va anar minvant a Europa i especialment a la regió mediterrània (Blondel & Aronson 1999), i les espècies associades a ambients agrícoles que ara estan en declivi, en aquest interval deurien augmentar de forma molt important. Per tant, és interessant tenir en compte que les prioritats de conservació van canviant amb el pas del temps, de manera lligada als canvis socioeconòmics que determinen la pressió que fa l'home sobre l'ambient on viu i les espècies amb les que coexisteix.

D'altra banda, l'efecte positiu de la tala post-incendi per a espècies amb interès de conservació s'ha d'emmarcar en aquest context mediterrani particular i en els canvis d'ús del sòl que s'hi han donat, comentats anteriorment. En altres parts del món, on la tala post-incendi també desencadena una substitució d'espècies forestals per altres de zones obertes (Morissette *et al.* 2002, Schwab *et al.* 2006), les particularitats i els interessos de conservació poden ser ben diferents. Aquest és el cas que es discuteix al segon capítol, en comparar l'efecte de la tala entre boscos de Catalunya i de l'oest de Montana. A les Muntanyes Rocalloses americanes, l'explotació intensa de l'hàbitat va començar fa tan sols dos segles, amb l'arribada dels colonitzadors europeus, i per tant estan molt menys alterades que la regió mediterrània, incloent-hi un paisatge menys

fragmentat amb boscos molt més continus. Els incendis forestals són, la gran majoria, d'origen natural (per llamps), cosa que constitueix una important diferència amb la regió mediterrània. A Nord Amèrica es considera que algunes espècies d'ocells han evolucionat amb els incendis forestals fins al punt de dependre dels boscos cremats, com és el cas del picot *Picoides arcticus* (Hutto 2008), que és molt més abundant als boscos cremats que en qualsevol altre hàbitat forestal. D'altra banda, la proporció d'ocells d'ambients oberts en zones cremades de les muntanyes Rocalloses és molt baixa, probablement per la llunyania de poblacions que podrien funcionar com a font (praderies, pastures o zones agrícoles). Tot i ser un grup d'ocells que també té interès de conservació (Brawn *et al.* 2001, Hunter *et al.* 2001), la tala post-incendi no les beneficia gaire. Per tant, als boscos de les Rocalloses, l'efecte de la tala post-incendi sobre els ocells és ben diferent d'aquell que té lloc a la regió mediterrània, on no hi ha ocells forestals que es beneficïïn (o depenguin) del foc. A diferència d'allò que succeeix als boscos de Montana, aquí la tala facilita la presència d'espècies amb interès de conservació. Per tant, tot i que els efectes de la tala siguin molt semblants pel que fa a les espècies beneficiades i perjudicades, agrupades pels seus requeriments ecològics, les recomanacions de gestió en un i altre lloc poden no ser comunes, i han de considerar la distribució a escala de paisatge de l'avifauna i les prioritats de conservació particulars de cada regió.

Tornant als boscos cremats catalans, és interessant destacar que també hi ha espècies d'ocells que no es veuen massa afectats per la desaparició dels pins cremats, sinó per altres aspectes de l'hàbitat que també canvien de manera important al llarg dels primers anys després del foc. Són les espècies més lligades a brolles, màquies i bardisses, (e.g. tots els tallarols del gènere *Sylvia*, i algunes altres), per a les quals és important comptar amb un mínim recobriment de l'estrat arbustiu (Pons *et al.* 2008). L'altra cara de la moneda és la dels ocells que necessiten sòl descobert, principalment per alimentar-se, com el pinsà comú *Fringilla coelebs*, la cardina *Carduelis carduelis* i el pardal comú *Passer domesticus*, als quals la regeneració vegetal els perjudica (capítol 1). D'altra banda, si la presència d'ocells de brolla ve determinada pel recobriment de l'estrat arbustiu, a d'altres ocells més típics de bardisses els afavoreix la presència de cursos d'aigua per colonitzar àrees cremades. En aquestes zones, després del foc la major humitat probablement afavoreixi una ràpida recuperació de la vegetació. Això permet la presència de rossinyols *Luscinia megarhynchos*, bosquetes *Hippolais polyglotta* i, fins i tot, ulls de bou *Troglodytes troglodytes*. Algunes espècies més pròpies d'ecotons, com el gratapalles *Emberiza cirlus*, també es beneficien d'aquesta vegetació propera a les rieres.

Els ocells frugívors i la dispersió de llavors

A la Conca Mediterrània els ocells frugívors són importants per a la dispersió de llavors de plantes amb fruit carnós (Herrera 1984), sobretot a la tardor i hivern; l'època quan la major part d'aquestes plantes presenten fruits madurs (Jordano 1992). Els resultats de

l'efecte de la tala sobre les espècies d'ocells a l'hivern és molt similar a tot el comentat fins ara per a la primavera, tal i com es mostra al capítol 3. Tot i que part de les espècies canvia respecte a l'època reproductora, fruit dels fluxos migratoris i dispersius, els ocells forestals segueixen ocupant les àrees no tallades, els d'espais oberts ocupen les que s'han tallat, i els de matollar, les zones on més recobriment ofereixi la brolla o el sotabosc (tot i preferir les que menys arbres tinguin). Tanmateix, el resultat més rellevant és que espècies frugívores seleccionen positivament els pilons de branques i feixines construïts en algunes zones després de tallar els pins cremats. El propòsit inicial d'aquests pilons és el d'actuar com a barreres contra l'erosió, per intentar minimitzar l'escorrentia en una situació de desprotecció del sòl. Els ocells frugívors utilitzen els pilons de branques –com a punts de guaita i d'alimentació-, més que cap altra estructura de l'hàbitat (terra, estrat arbustiu, tronc cremat o pi viu). A més, zones amb més pilons alberguen més ocells frugívors i de més espècies.

Figura 3. El tallarol capnegre *Sylvia melanocephala* és l'ocell frugívor i dispersor de llavors més abundant de les zones cremades de terra baixa mediterrània.

Les conseqüències d'aquesta selecció del microhàbitat per a la dispersió de llavors per part dels ocells frugívors es posen de manifest al capítol 4: la pluja de llavors és molt més elevada sota aquestes feixines que en qualsevol de les altres estructures estudiades (sobre el terra, sota l'estrat arbustiu, sota arbres aïllats o sota arbres agrupats), i és equiparable a la dispersió que té lloc en boscos veïns no cremats. Aquest resultat contrasta amb el que s'esperava dels arbres aïllats, considerats a priori com a importants per als ocells dispersors, sobre els quals tindrien un hipotètic *perch effect*, un efecte d'atracció o reclam que ha estat trobat per altres autors, tot i que no en condicions post-incendi (Guevara & Laborde 1993, Pausas *et al.* 2006). Podria ser que per als dispersors de costums més o menys arbòries (el tord *Turdus philomelos* i, potser en menor mesura, la merla *Turdus merula*), els arbres cremats resultin poc atractius, cosa que es podia esperar més de dispersors que seleccionen estrats de vegetació més baixos (tallarols, pit-roig), sobre els quals els pilons sí que exerceixen aquesta atracció.

En qualsevol cas, un resultat no menys important del capítol 4 és que els ocells frugívors poden dispersar llavors de plantes amb fruit carnós consumides a l'exterior del perímetre cremat cap a dins d'aquest, un procés no descrit amb anterioritat. Això confirma que la dispersió per ocells frugívors és un mitjà de colonització d'àrees cremades per part d'espècies vegetals amb fruit carnós, la qual cosa ja havia estat suggerida per Cowling *et al.* (1997). Així mateix, alguns mamífers carnívors, com les guilles *Vulpes vulpes*, gorjablancs *Martes foina* i teixons *Meles meles*, s'ha vist que també són responsables de la dispersió de llavors de fruits carnosos cap a l'interior de zones incendiades (resultats inèdits). La magnitud d'aquesta dispersió no havia estat valorada, que en tinguem coneixement, fins aquesta tesi. Cal dir, però, que en aquest estudi (capítol 4) no es va trobar dependència entre l'abundància de llavors dispersades i la distància a les possibles fonts de colonització, situades fora del perímetre cremat. Estudis previs sobre les distàncies de dispersió de llavors per ocells a l'estiu (Jordano *et al.* 2007) situaven el límit de dispersió dels ocells petits (tallarols) vora els 100 m de la planta mare i dels mitjans (un túrdid, la griva *Turdus viscivorus*) fins al mig quilòmetre. En el capítol 4, la font de llavors més propera es situava a uns 200 m de la unitat mostral més propera, la qual cosa podria suggerir que en aquest cas la dispersió es deu principalment a ocells de mida mitjana (com podrien ser, a l'àrea d'estudi de la tesi, la merla i el tord). El fet que una bona part de els llavors recollides sota els pilons fossin d'olivera *Olea europaea*, també aniria en aquesta direcció.

Als capítols 1 i 3 es mostra com aquests túrdids no mostren cap associació particular amb les variables ambientals mesurades a les zones cremades i gestionades, com també van trobar Izhaki & Adar (1997). Això sugereix que la tala no els hauria de perjudicar especialment, així com tampoc al procés de dispersió de llavors que duen a terme a l'hivern, atès que no són les zones amb arbres (ja siguin peus aïllats o zones no internvingudes) on la pluja de llavors és més intensa, sinó a les feixines de branques cremades que es troben a la zona talada. De totes maneres, en un estudi fet a Sierra Nevada, Castro *et al.* (2010) van trobar que els túrdids i el gaig *Garrulus glandarius* – important dispersor d'aglans- eren més abundants en zones no talades, resultats que contrasten amb les associacions més aviat febles, i no significatives, d'aquestes espècies amb ambients forestals a terra baixa (capítols 1 i 3). Probablement, la més ràpida regeneració vegetal a baixa altitud provoca que els dispersors, especialment els túrdids, puguin ocupar una varietat d'ambients més elevada que no en ambients de muntanya, després del foc. Això, tenint en compte que les merles ocupen principalment el sotabosc i els tords no són rars a la brolla, a l'hivern. En qualsevol cas, la presència de dispersors i la dispersió de llavors que duen a terme són aspectes condicionats també per qüestions d'escala (Peterson *et al.* 1998, Kollmann 2000). A les àrees forestals cremades on es van dur a terme aquests treballs, de superfície relativament poc extensa (de 613 ha la més gran) en comparació amb incendis forestals de milers d'hectàrees, la proximitat al marge de l'incendi i de taques de bosc no cremades ni talades al seu interior, podria haver facilitat la presència d'aquests dispersors més forestals dins la zona talada. Restaria per estudiar, per tant, si també són presents en zones cremades i talades més àmplies.

Figura 4. Els pilons de branques són estructures que afavoreixen la presència d'ocells frugívors, reben una pluja de llavors important i presenten abundàncies de plantes dispersades per ocells més elevades que a la vegetació dels voltants. En aquesta fotografia aèria es poden observar la quantitat de feixines (línies fosques discontínues que segueixen les corbes de nivell) que es van construir per a evitar l'erosió en moltres parts de l'incendi de la Vall de Santa Caterina, a Torroella de Montgrí (cremat el 2004, fotografia presa dos anys més tard; Font: Google Earth).

Els resultats que mostraven una major abundància d'ocells frugívors a les feixines, i una pluja de llavors més intensa sota les mateixes, van conduir a pensar si realment aquesta dispersió es traduïa en més plantes amb fruit carnós als pilons que al seu voltant. Al capítol 5 els resultats mostren com això es compleix en determinades zones, que van coincidir amb els vessants més assolellats i secs (els orientats al sud), on la regeneració de la coberta vegetal és més lenta després del foc (Raftoyannis & Spanos 2005). Els resultats dels dos capítols anteriors podrien suggerir que l'efecte positiu dels pilons es degui a una major dispersió per ocells als pilons. De totes maneres, no es pot assegurar que les plantes amb fruit carnós d'aquestes àrees tinguin el seu origen en la dispersió; per contra, bona part de les espècies també poden regenerar per rebrot, després del foc (Pausas & Verdú 2005). El que resta fora de dubte és l'efecte positiu que tenen les feixines i pilons per a aquestes plantes, essent per tant un microhàbitat de qualitat per a la regeneració de les plantes estudiades, sigui quin sigui el seu origen. Aquesta qualitat podria estar relacionada amb unes millors condicions microclimàtiques per a les plàntules als pilons que fora d'ells, en proporcionar aquestes estructures més ombra i més humitat que un ambient descobert (Bautista *et al.* 2004, Castro *et al.* 2008), condicions que podrien ser més accentuades en pilons més grans. Això podria suposar un avantatge per a la supervivència dels plançons, que és més elevada sota els pilons per a espècies com el ginebre *Juniperus communis* (Castro *et al.* 2008). Per altra banda, l'estructura de la feixina pot tenir també un efecte protector davant de la depredació de

plàntules i granes per part de mamífers mitjans i grans (Puerta-Piñero *et al.* 2010), tot i que davant de rosegadors l'efecte podria ser més limitat o fins i tot contraproduent, cosa que resta per valorar. Tot i això, la depredació no sembla ser un problema en zones on la pluja de llavors es dóna en quantitats elevades i els depredadors no estan limitats per aquest recurs (Calviño-Cancela 2007). Els factors que fan dels pilons de branques bons microhàbitats per a les plantes amb fruit carnós, en tot cas, podrien variar d'espècie en espècie, perquè a algunes els afecta només la presència de pilons (e.g. la rogeta *Rubia peregrina*), mentre que a d'altres també els afecta el lloc i l'alçada del piló (e.g. l'esparraguera *Asparagus acutifolius*).

Aquests resultats sobre la dispersió per endozoocòria de plantes amb fruit carnós cap a àrees cremades, i la influència que hi té la gestió post-incendi, són rellevants en el marc de la recuperació de la vegetació després del foc. La majoria d'espècies de plantes mediterrànies tenen estratègies que els permeten regenerar *in situ*, per rebrotada i/o per germinació a partir d'un banc de llavors (Pausas & Verdú 2005), en un procés anomenat autosuccessió que fa que la vegetació mediterrània sigui molt resilient al foc (Trabaud & Lepart 1980; Lavorel 1999). Des d'un punt de vista quantitatiu (de recobriment vegetal), la contribució relativa d'aquesta colonització des de l'exterior a la recuperació de la vegetació podria ser força limitada, en comparació a les estratègies que permeten l'autosuccessió. No obstant, els mecanismes de dispersió i colonització són claus per a la dinàmica de les poblacions de plantes (Rey & Alcántara 2000), influïnt, per exemple, en l'estructura genètica de les mateixes (Loveless & Hamrick 1984).

D'altra banda, els pilons de branques han resultat una estructura clau per a la presència d'aquestes plantes dispersades per ocells en determinades àrees. De totes maneres, seria molt interessant analitzar si això es deu únicament a una major dispersió de llavors, o si s'hi combina també un possible efecte de facilitació (e.g. per unes millors condicions ambientals sota els pilons). En aquest sentit es podria analitzar si altres plantes, més enllà de les dispersades per ocells, són també més abundants en pilons que en altres zones, en els vessants més secs. En qualsevol cas, els resultats del capítol 5 posen de manifest que la presència de pilons de branques provoca que les plantes amb fruit carnós apareguin en llocs on no hi serien en absència d'aquestes estructures. Finalment, els resultats d'aquests capítols són també rellevants per a la gestió, atès que mostren que la construcció de pilons i feixines de branques pot ser una eina molt interessant en la recuperació d'espècies amb fruit carnós a les parts més àrides de les àrees cremades.

Propostes de gestió post-incendi

Els resultats i conclusions que es presenten en aquesta tesi tenen, a més d'un interès purament ecològic, una rellevància pràctica, atès que aporten informació nova que pot ser tinguda en consideració a l'hora de decidir com gestionar els boscos de pi blanc cremats a Catalunya i en altres parts de la Conca Mediterrània de característiques similars. En aquesta tesi es mostra com l'aprofitament de la fusta cremada pot ser

compatible amb la conservació d'espècies d'ocells en declivi i de processos d'interacció entre fauna i vegetació. Tanmateix, els criteris de conservació no solen regir les activitats de gestió post-incendi, almenys quan es duen a terme amb finalitats purament comercials, i en els casos de boscos públics on els criteris ambientals poden tenir més pes, l'escassetat d'informació disponible sobre quins tractaments són més adequats, basada en estudis científics, és sempre un problema. Els resultats dels treballs que es presenten en aquesta tesi poden contribuir, per tant, a omplir part d'aquest buit d'informació pel que fa a l'avifauna i a la vegetació. A continuació es comenten algunes propostes de gestió que es podrien implementar en futurs plans de gestió de boscos cremats, basades en el que s'exposa en els diferents capítols d'aquesta tesi.

Un dels resultats principals dels treballs presentats és el benefici que obtenen els ocells propis d'espais oberts, molts d'ells en situació de conservació desfavorable, de la tala dels pins cremats. En principi, per tant, la tala post-incendi seria una activitat prou compatible amb la conservació d'aquestes espècies. Tanmateix, moltes d'aquestes espècies necessiten també estructures verticals com a punts elevats de guaita, o com a llocs de cant, o de descans (en serien casos el cruixidell, l'abellerol *Merops apiaster*, el capsigrany *Lanius senator*, el botxí *Lanius meridionalis*, el cotoliu *Lullula arborea*, el passerell *Carduelis cannabina*, el xoriguer *Falco tinnunculu*, ...). Els arbres cremats fan aquesta funció, de manera que la seva desaparició completa després de la tala pot resultar perjudicial per a espècies que en un principi s'havien vist afavorides per un hàbitat més obert. Per tant, en els llocs on es tallin els arbres seria interessant no endur-se'ls tots, sinó deixar-ne alguns de drets. Quines característiques han de tenir els arbres cremats que no es tallin no és una qüestió fàcil de respondre, perquè si bé els individus més joves –menys valuosos comercialment- ja podrien complir aquesta funció, els més vells poden oferir recursos que esdeven escassos amb la tala, com les cavitats. Tampoc ho és la densitat amb què s'haurien de deixar aquests arbres, quedant pendent per a futures investigacions, però no hauria de ser necessàriament alta. D'altra banda, mantenir rodals sense talar també podria evitar la desaparició d'ocells amb requeriments forestals, alguns dels quals també tenen una situació de conservació desfavorable, com la tórtora i la mallerenga emplomallada. Altres espècies (tord, merla, gaig...) que poden ser importants per al procés de dispersió de llavors, en determinades àrees, són més abundants a les zones no talades, de manera que aplicant un principi de precaució, el millor podria ser deixar també alguns rodals sense intervenció. Per tant, probablement el model de gestió de pinedes cremades més adequat és una gestió mixta, talant els pins d'una part de la zona cremada (cosa que d'altra banda permetria obtenir un cert rendiment econòmic) però mantenint una mínima densitat de peus cremats drets, i deixant de talar alguns rodals per afavorir la presència d'espècies més forestals que també poden ser objectiu de conservació.

D'altra banda, els tres darrers capítols de la tesi posen de manifest la importància de construir pilons de branques, per afavorir els ocells frugívors, per al propi procés de dispersió de llavors i per a la distribució de plantes dispersades per ocells. L'objectiu primer d'aquests pilons o feixines és el de fer de barreres per a l'escorrentia, reduint així

l'erosió del sòl (McIver & Starr 2000). Aquest efecte, tanmateix, ha estat posat en entredit per alguns estudis (Wohlgemuth *et al.* 2001, Marquès & Mora 1998), i s'ha vist que aquestes barreres són poc efectives davant d'episodis de precipitació intensa (Robichaud *et al.* 2008), que no són estranyes a la regió mediterrània. Tot i això, els seus efectes sobre la biodiversitat i sobre el manteniment de processos ecològics com la dispersió de llavors són clarament beneficiosos, i es tracta, per tant, d'un tractament de gestió post-incendi molt recomanable, independentment que sigui efectiu o no per a reduir l'erosió. Les feixines contra l'erosió se solen construir seguint les corbes de nivell i sovint emprant tot l'arbre cremat, fixant el tronc a terra amb estaques i apilant-hi a sobre les branques més primes (Robichaud *et al.* 2008). Aquestes estructures, si són prou altes (entre 0.5 i 1 m d'alçada, capítols 3 i 5) i es construeixen en prou densitat (15-25 pilons/ha, capítol 3), van bé per als ocells i la vegetació. Fins i tot pilons més senzills, construïts només amb les branques més primes, però amb l'alçada i densitat adequades, probablement fossin igual d'eficaços, i permetrien aprofitar comercialment la fusta dels troncs gruixuts. A més d'ocells, els pilons de branques també beneficien alguns rèptils (Santos & Poquet 2010), micromamífers (Haim & Izhaki 1994) i el conill, una espècie clau dels ecosistemes mediterranis (Rollán & Real 2010).

Aquestes recomanacions de gestió no serien tampoc contràries a la conservació del sòl o la regeneració de la vegetació, dos aspectes que també poden veure's afectats per les activitats derivades de la tala post-incendi. L'erosió és un aspecte preocupant després d'un incendi, per la desprotecció del sòl davant d'episodis de precipitació intensa, i que es podria veure agreujat per l'aixafament de vegetació i pertorbació del sòl a resultes del trànsit amb maquinària pesada (e.g. *skidders*, remolcs...) per a dur a terme els treballs forestals (McIver & Starr 2000, Beschta *et al.* 2004). L'efecte d'aquests treballs s'ha quantificat en un augment del 15% en l'erosió respecte de llocs no gestionats, tot i que les diferències es van reduint a mesura que el recobriment de la vegetació augmenta, i es fan imperceptibles a mig terme, una dècada després del foc (Bautista *et al.* 2004, Spanos *et al.* 2005, Fernández *et al.* 2007). Tanmateix, sí que es produeix un increment persistent en l'erosió en els casos en què les restes no aprofitables comercialment no es preserven *in situ* sinó que es cremen, en vessants amb substrats susceptibles a perdre sòl, com els sorrencs, i quan s'obren pistes forestals en llocs de fort pendent (Marquès & Mora 1998, Bautista *et al.* 2004). Per altra part, l'efecte dels treballs relacionats amb tala post-incendi sobre la regeneració de la vegetació es considera poc important (Ne'eman *et al.* 1995), i els efectes negatius a curt termini també van desapareixent en pocs anys (Martínez-Sánchez *et al.* 1999, Bautista 2004, Spanos *et al.* 2005, Raftoyannis & Spanos 2005). Per tant, aquests resultats indicarien que els efectes de la tala post-incendi sobre la recuperació de l'ecosistema no han d'implicar greus perjudicis a mig i llarg termini per a la conservació del sòl i la recuperació de la vegetació, sempre i quan es tinguin en compte accions senzilles i poc costoses econòmicament com evitar l'obertura de pistes en zones de pendent fort, no realitzar tales en zones de sòls vulnerables a l'erosió, i deixar les restes no aprofitables sobre el terreny enlloc de cremar-les, preferentment amuntegades en pilons.

Les recomanacions de gestió basades en els resultats d'aquesta tesi, per tant, indiquen que una gestió post-incendi mixta permetria compaginar l'aprofitament comercial d'una part dels recursos fustaners amb la conservació d'espècies d'ocells amb interès de conservació i de processos importants per a l'ecosistema com la dispersió de llavors. Per tant, seria interessant incloure aquestes propostes a les directrius i estratègies de gestió post-incendi en boscos de titularitat pública, on la gestió és responsabilitat de l'administració, i també a les condicions necessàries per a la concessió de subvencions encaminades a la gestió de finques privades. D'altra banda, cal també destacar la conveniència de continuar la recerca sobre els efectes de la gestió post-incendi sobre l'avifauna, per tal de perfeccionar-ne el coneixement, i ampliar-la a aquells grups i processos en els que encara existeixen buits d'informació, per tal que les decisions pràctiques de gestió es basin sempre que sigui possible en evidències científiques ben contrastades.

Conclusions

1. La tala post-incendi és el principal factor que determina la composició de la comunitat d'ocells en pinedes cremades mediterrànies, facilitant la substitució d'espècies forestals per espècies pròpies d'espais oberts.

2. Com a resultat de la tala post-incendi, el valor de conservació de la comunitat d'ocells s'incrementa, a causa que bona part dels ocells d'espais oberts que es veuen beneficiats per la tala presenten una situació de conservació desfavorable a escala europea.

3. L'efecte negatiu de la tala post-incendi sobre ocells forestals és un patró general en comunitats d'ocells de regions diferents, com la Conca Mediterrània i les muntanyes Rocalloses americanes. Tanmateix, les diferències entre l'avifauna d'ambdues regions aconsellen mesures de gestió diferents.

4. Els pilons de branques cremades construïts com a feixines contra l'erosió afavoreixen la presència d'ocells frugívors a l'hivern, que els utilitzen més que cap altra estructura resultant de la gestió forestal post-incendi, com a punts d'alimentació, de guaita o de refugi.

5. Els ocells frugívors dispersen llavors de plantes amb fruit carnós cap a l'interior de pinedes cremades i gestionades. La pluja de llavors d'aquestes plantes derivada de les deposicions dels ocells és més intensa als pilons de branques que en cap altra estructura resultant de la gestió post-incendi.

6. Les espècies de plantes amb fruit carnós dispersades per ocells frugívors són més abundants en pilons de branques cremades que fora d'aquestes estructures, especialment als pilons de més alçada. Aquest efecte és més evident als vessants més secs, la qual cosa suggereix que els pilons poden presentar condicions

microclimàtiques especialment favorables per a l'establiment d'aquestes plantes en aquestes àrees.

7. La tala post-incendi és una activitat en bona part compatible amb la conservació d'ocells d'espais oberts a la regió d'estudi, tot i que seria interessant deixar arbres cremats aïllats sense tallar com a punts de guaita, cant o descans per aquestes espècies, i deixar també alguns rodals sense tallar per mantenir espècies forestals amb interès de conservació. Aquests rodals podrien estar a les àrees més vulnerables a l'erosió on la tala pot suposar un risc més elevat de pèrdua de sòl.

8. La construcció de pilons de branques amb les restes de tala no aprofitables comercialment, com a feixines contra l'erosió, és una mesura de gestió altament recomanable per mantenir processos com la dispersió de llavors per part d'ocells frugívors, i per afavorir la presència d'una bona quantitat d'espècies de fauna que aprofiten també aquestes estructures.

Referències

Barlow, J. & Peres, C.A. 2004. Avifaunal responses to single and recurrent wildfires in Amazonian forests. Ecological Applications 14, 1358-1373.

Bautista, S., Gimeno, T., Mayor, G.A. & Gallego, D. 2004. El tratamiento de la madera quemada tras los incendios forestales. A: Vallejo, R.A. & Alloza, J.A. (eds.). Avances en el estudio de la gestión del monte mediterráneo. Fundación Centro de Estudios Ambientales del Mediterráneo (CEAM), València. Pp 547-570.

Beschta, R.L., Rhodes, J.J., Kauffman, J.B., Gresswell, R.E., Minshall, G.W., Karr, J.R., Perry, D.A. Hauer, F.R. & Frissell, C.A. 2004. Postfire management on forested public lands of the western United States. Conservation Biology 18, 957-967.

BirdLife International. 2004. Birds in Europe: population estimates, trends and conservation status. BirdLife International, Wageningen.

BirdLife International. 2008. *Sylvia undata*. A: IUCN 2010. IUCN Red List of Threatened Species. Version 2010.4. www.iucnredlist.org.

Blondel, J. & Aronson, J. 1999. Biology and wildlife of the Mediterranean region. Oxford University Press, New York.

Brawn, J.D., Robinson, S.K., Thompson III, F.R. 2001. The role of disturbance in the ecology and conservation of birds. Annual Review of Ecology and Systematics 32, 251-276.

Brotons, L. Pons, P. & Herrando, S. 2005. Colonization of dynamic Mediterranean landscapes: where do birds come from after fire? Journal of Biogeography 32, 789-798.

Brotons, L., Herrando, S. & Pons, P. 2008. Wildfires and the expansion of threatened farmland birds: the ortolan bunting *Emberiza hortulana* in Mediterranean landscapes. Journal of Applied Ecology 45, 1059-1066.

Calviño-Cancela, M. 2007. Seed and microsite limitations of recruitment and the impacts of post-dispersal seed predation at the within population level. Plant Ecology 192, 35-44.

Castro, J., Sánchez-Miranda, A., Lorite, J. & Zamora, R. 2008. Resultados preliminares del efecto de los tratamientos selvícolas post-incendio relacionados con la madera quemada sobre el establecimiento de plantones de *Juniperus communis* l. en laalta montaña mediterránea. Cuadernos de la Sociedad Española de Ciencias Forestales 28, 31-36.

Castro, J., Moreno-Rueda, G. & Hódar, J.H. 2010. Experimental test of post-fire management in pine forests: impact of salvage logging versus partial cutting and non-intervention on bird-species assemblages. Conservation Biology 24, 810-819.

Clavero, M. & Brotons, L. 2010. Functional homogenization of bird communities along habitat gradients: accounting for niche multidimensionality. Global Ecology and Biogeography 19, 684-696.

Cowling, R.M., Kirkwood, D., Midgley, J.J., Pierce, S.M., 1997. Invasion and persistence of bird-dispersed, subtropical thicket and forest species in fire-prone coastal fynbos. Journal of Vegetation Science 8, 475-488.

Donald, P.F., Green, R.E. & Heath, M.F. 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. Proceedings of the Royal Society of London 268, 25-29.

Estrada, J., Pedrocchi, V., Brotons, L. & Herrando, S. (eds.) 2004. Atles dels ocells nidificants de Catalunya 1999-2002. Institut Català d'Ornitologia (ICO) / Lynx Edicions. Barcelona.

Fernández, C., Vega, J.A., Fonturbel, T., Pérez-Gorostiaga, P., Jiménez, E. & Madrigal, J. 2007. Effects of wildfire, salvage logging and slash treatments on soil degradation. Land Degradation & Development 18, 591-607.

Guevara, S., Laborde, J., 1993. Monitoring seed dispersal at isolated standing trees in tropical pastures – consequences for local species availability. Vegetatio 108, 319-338.

Haim, A. & Izhaki, I. 1994. Changes in rodent community during recovery from fire: relevance to conservation. Biodiversity and Conservation 3, 573-585.

Herrando, S., Brotons, L., Del Amo, R. & Llacuna, S. 2002. Bird community succession after fire in a dry Mediterranean shrubland. Ardea 90, 303-310.

Herrando, S., Brotons, L. & Llacuna, S. 2003. Does fire increase the spatial heterogeneity of bird communities in Mediterranean landscapes? Ibis 145, 307-317.

Herrera, C.M. 1984. A study of avian frugivores, bird-dispersed plants, and their interaction in Mediterranean scrublands. Ecological Monographs 54, 1-23.

Hessburg, P.F. & Agee, J.K. 2003. An environmental narrative of Inland Northwest United States forests, 1800–2000. Forest Ecology and Management 178, 23-59.

Hunter, W.C., Buehler, D.A., Canterbury, R.A. Confer, J.L. & Hamel, P.B. Conservation of disturbance-dependent birds in eastern North America. Wildlife Society Bulletin 29, 440-455.

Hutto, R.L. 1995. Composition of bird communities following stand-replacement fires in northern Rocky Mountain (U.S.A.) conifer forests. Conservation Biology 9, 1043-1058.

Hutto, R.L. 2008. The ecological importance of severe wildfires: some like it hot. Ecological Applications 18, 1827-1834.

ICO. 2010. Vuitè informe del Programa de Seguiment d'Ocells Comuns a Catalunya (SOCC). Institut Català d'Ornitologia, Barcelona.

IUCN. 2001. IUCN Red List Categories and Criteria: Version 3.1. IUCN, Gland i Cambridge.

IUCN. 2003. Guidelines for application of IUCN Red List Criteria at regional levels: Volume 3.0. IUCN, Gland i Cambridge.

Izhaki, I. & Adar, M. 1997. The effects of postfire management on bird community succession. International Journal of Wildland Fire 7, 335-342.

Jacquet, K. & Prodon, R. 2009. Measuring the postfire resilience of a bird-vegetation system: a 28-year study in a Mediterranean oak woodland. Oecologia 161, 801-811.

Jordano, P. 1992. Fruits and frugivory. In: Fenner, M. (Ed.), Seeds. The Ecology of Regeneration in Plant Communities. C.A.B. International, Wallingford. Pp. 125-166.

Jordano, P., García, C., Godoy, J.A. & García-Castaño, J.L. 2007. Differential contribution of frugivores to complex seed dispersal patterns. Proceedings of the National Academy of Sciences of the United States of America 104, 3278-3282.

Karr, J.R. & Freemark, K.E. 1985. Disturbance and vertebrates: an integrative perspective. En: Pickett, S.T.A. & White, P.S. The ecology of natural disturbance and patch dynamics. Academic Press, San Diego. Pp. 153-168.

Kollmann, J. 2000. Dispersal of fleshy-fruit species: a matter of spatial scale? Perspectives in Plant Ecology, Evolution and Systematics 3, 29-51.

Kotliar, N.B., Hejl, S.J., Hutto, R.L., Saab, V., Melcher, C.P. & McFadzen, M.E. 2002. Effects of fire and postfire salvage logging on avian communities in conifer dominated forests of the western United States. Studies in Avian Biology 25, 49-64.

Lavorel, S. 1999. Ecological diversity and resilience of Mediterranean vegetation to disturbance. Diversity and Distributions 5, 3-13.

Llimona, F., Matheu, E., Prodon, R., 1993. Role of snag persistence and of tree regeneration in postfire bird successions: comparison of pine and oak forests in Montserrat (Catalonia, NE Spain). A: Trabaud, L., Prodon, R. (eds.). Fire in Mediterranean Ecosystems. Ecosystems Research Report, 5. Commission of the European Communities, Brussel·les. Pp. 315–331.

Loveless, M. & Hamrick, J. 1984. Ecological determinants of genetic structure in plant populations. Annual Review in Ecology Evolution and Systematics 15, 65–95.

Marquès, M.A. & Mora, E. 1998. Effects on erosion of two post-fire management practices: clear-cutting versus non-intervention. Soil & Tillage Research 45, 433-439.

Martí, R. & del Moral, J.C. (eds.). 2003. Atlas de las aves reproductoras de España. Dirección General de Conservación de la Naturaleza-Sociedad Española de Ornitología, Madrid

Martínez-Sánchez, J.J., Ferrandis, P., de las Heras, J. & Herranz, J.M. 1999. Effect of burnt wood on the natural regeneration of Pinus halepensis after fire in a pine forest in Tus Valley (SE Spain). Forest Ecology and Management 123, 1-10.

McIver, J.D. & Starr, L. 2000. Environmental effects of postfire logging: literature review and annotated bibliography. General Technical Report PNW-GTR-486. USDA Forest Service, Portland.

Morissette, J.L., Cobb, T.P., Brigham, R.M. & James, P.C., 2002. The response of boreal songbird communities to fire and postfire harvesting. Candian Journal of Forest Research 32, 2169-2183.

Ne'eman, G., Lahav, H. & Izhaki, I. 1995. Recovery of vegetation in a natural east Mediterranean pine forest on Mount Carmel, Israel as affected by management strategies. Forest Ecology and Management 75, 17-26.

Pausas, J.G. & Verdú, M. 2005. Plant persistence traits in fire-prone ecosystems of the Mediterranean basin: a phylogenetic approach. Oikos 109, 196-202.

Pausas, J.G., Bonet, A., Maestre, F.T., Climent, A. 2006. The role of the perch effect on the nucleation process in Mediterranean semi-arid oldfields. Acta Oecologica 29 346-352.

PECBMS. 2009. The state of Europe's common birds 2008. CSO/RSPB, Praga.

Peterson, G., Allen, C.R. & Holling, C.S. Ecological resilience, biodiversity, and scale. Ecosystems 1, 6-18.

Pons, P. & Bas, J.M. 2005. Open-habitat birds in recently burned areas: the role of the fire extent and species' habitat breadth. Ardeola 52, 119-131.

Pons, P., Bas, J.M., Prodon, R., Roura-Pascual, N. & Clavero, M. 2008. Territory characteristics and coexistence with heterospecifics in the Dartford warbler *Sylvia undata* across a habitat gradient. Behavioral Ecology and Sociobiology 62, 1217-1228.

Pons, P. & Clavero, M. 2010. Bird responses to fire severity and time since fire in managed mountain rangelands. Animal Conservation 13, 294-305.

Preiss, E., Martin, J.L. & Debussche, M. 1997. Rural depopulation and recent landscape changes in a Mediterranean region: Consequences to the breeding avifauna. Landscape Ecology 12, 51-61.

Puerta-Piñero, C., Sánchez-Miranda, A., Leverkus, A. & Castro, J. 2010. Management of burnt wood after fire affects post-dispersal acorn predation. Forest Ecology and Management 260, 345-352.

Raftoyannis, Y. & Spanos, I. 2005. Evaluation of log and branch barriers as post-fire rehabilitation treatments in a Mediterranean pine forest in Greece. International Journal of Wildland Fire 14, 183-188.

Rey, P.J. & Alcántara, J.M. 2000. Recruitment dynamics of a fleshy-fruited plant (*Olea europaea*): connecting patterns of seed dispersal to seedling establishment. Journal of Ecology 88, 622-633.

Robichaud, P.R., Wagenbrenner, J.W., Brown, R.E., Wohlgemuth, P.M. & Beyers, J.L. 2008. Evaluating the effectiveness of contour-felled logs erosion barriers as a post-fire runoff and erosion mitigation treatment in the western United States. International Journal of Wildland Fire 17, 255-273.

Rollán, A. & Real, J. 2010. Effect of wildfires and post-fire forest treatments on rabbit abundance. European Journal of Wildlife Research, in press. DOI: 10.1007/s10344-010-0412-y.

Santos, X. & Poquet, J.M. 2010. Ecological succession and hábitat attributes affect the postfire response of a Mediterranean reptile community. European Journal of Wildlife Research 56, 895-905.

Schwab, F.E., Simon, N.P.P., Stryde, S.W. & Forbes, G.J. 2006. Effects of postfire snag removal on breeding birds of western Labrador. Journal of Wildlife Management 70, 1464-1469.

Sirami, C., Brotons, L. & Martin, J.L. 2007. Vegetation and songbird response to land abandonment: from landscape to census plot. Diversity and Distributions 13, 42-52.

Smucker, K.M., Hutto, R.L. & Steele, B.M. 2005. Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications 15, 1535-1549.

Snow, D. & Perrins, C. (eds.). 1998. The complete Birds of the Western Palearctic on CD-ROM. Oxford University Press, Oxford.

Spanos, I., Raftoyannis, Y., Goudelis, G., Xanthopoulou, E., Samara, T. & Tsiontsis, A. 2005. Effects of postfire logging on soil and vegetation recovery in a Pinus halepensis Mill. forest in Greece. Plant and Soil 278, 171-179.

Trabaud, L., Lepart, J., 1980. Diversity and stability in garrigue ecosystems after fire. Vegetatio 43, 49-57.

Traveset, A., & Riera, N. 2005. Disruption of a plant-lizard seed dispersal system and its ecological effects on a threatened endemic plant in the Balearic Islands. Conservation Biology 19, 421-431.

Ukmar, E., Battisti, C., Luiselli, L. & Bologna, M.A. 2007. The effects of fire on communities, guilds and species of breeding birds in burnt and control pinewoods in central Italy. Biodiversity and Conservation 16, 3287-3300.

Vorisek, P., Gregory, R.D., Van Strien, A.J. & Gmelig, A. 2008. Population trends of 48 common terrestrial bird species in Europe: results from the Pan-European Common Bird Monitoring Scheme. Revista Catalana d'Ornitologia 24, 4-14.

Wohlgemuth, P.M., Hubbert, K.R. & Robichaud, P.R. 2001. The effects of log erosion barriers on post-fire hydrologic response and sediment yield in smal forested watersheds, southern California. Hydrological Processes 15, 3053-3066.

Wright, S.J. & Duber, H.C. 2001. Poachers and forest fragmentation alter seed dispersal, seed survival, and seedling recruitment in the palm *Attalea butyraceae*, with implications for tropical tree diversity. Biotropica 33, 583-595.

Annexos

Annex 1 (capítol 1)

Table A1.1. List of species found during the study and the number of individuals of each one detected in each year. We also show each species' habitat group, according to the regional EBCC (2010) classification.

Scientific name	Common name	Habitat	Detected individuals			
			2007	2008	2009	total
Accipiter gentilis	Goshawk		1	0	0	1
Accipiter nisus	Sparrowhawk	Forest	1	0	0	1
Aegithalos caudatus	Long-tailed tit	Forest	10	0	0	10
Anthus campestris	Tawny pipit	Farmland	1	2	9	12
Buteo buteo	Common buzzard		1	2	1	4
Caprimulgus europaeus	Nightjar		1	1	0	2
Carduelis cannabina	Linnet	Farmland	26	16	30	72
Carduelis carduelis	Goldfinch	Farmland	29	14	15	58
Carduelis chloris	Greenfinch	Farmland	18	8	18	44
Certhia brachydactyla	Short-toed treecreeper	Forest	17	12	3	32
Clamator glandarius	Great spotted cuckoo		0	1	0	1
Circaetus gallicus	Short-toed snake-eagle		0	0	2	2
Cisticola juncidis	Zitting cisticola		5	0	6	11
Columba palumbus	Wood pigeon		11	6	6	23
Cuculus canorus	Common cuckoo		3	1	0	4
Dendrocopos major	Great spotted woodpecker	Forest	4	7	3	14
Emberiza cirlus	Cirl bunting	Farmland	7	7	14	28
Erithacus rubecula	European robin		17	6	2	25
Falco subbuteo	Hobby		0	0	3	3
Falco tinnunculus	Common kestrel	Farmland	1	4	0	5
Fringilla coelebs	Chaffinch	Forest	27	8	3	38
Galerida cristata	Crested lark	Farmland	6	7	16	29
Garrulus glandarius	European jay		10	11	5	26
Hippolais polyglotta	Melodious warbler		2	7	30	39
Lanius collurio	Red-backed shrike		0	0	1	1
Lanius meridionalis	Southern grey shrike		0	0	3	3
Lanius senator	Woodchat shrike	Farmland	2	19	22	43
Lullula arborea	Wood lark	Farmland	23	36	58	117
Luscinia megarhynchos	Common nightingale	Farmland	17	19	10	46
Merops apiaster	Bee-eater	Farmland	16	12	29	57
Miliaria calandra	Corn bunting	Farmland	2	17	46	65
Motacilla alba	White wagtail	Farmland	3	1	1	5
Muscicapa striata	Spotted flycatcher	Forest	8	5	4	17
Oriolus oriolus	Golden oriole		5	7	1	13
Parus caeruleus	Blue tit	Forest	11	2	1	14
Parus cristatus	Crested tit	Forest	24	16	13	53
Parus major	Great tit		22	26	3	51
Passer domesticus	House sparrow	Farmland	17	10	10	37
Phylloscopus bonelli	Bonelli's warbler	Forest	1	3	0	4
Petronia petronia	Rock sparrow	Farmland	1	0	5	6
Pernis apivorus	Honey buzzard		0	0	1	1
Picus viridis	Green woodpecker	Forest	1	3	2	6
Regulus ignicapillus	Firecrest	Forest	1	0	0	1
Saxicola torquata	Common stonechat	Farmland	2	7	26	35

Scientific name	Common name	Habitat	Detected individuals			
			2007	2008	2009	total
Serinus serinus	Serin	Farmland	24	19	8	51
Streptopelia turtur	Turtle dove	Farmland	26	20	7	53
Sylvia atricapilla	Blackcap		8	3	0	11
Sylvia cantillans	Subalpine warbler		20	25	29	74
Sylvia melanocephala	Sardinian warbler		26	58	87	171
Sylvia undata	Dartford warbler		1	9	22	32
Sturnus vulgaris	Common starling	Farmland	0	2	0	2
Troglodytes troglodytes	Winter wren	Forest	7	6	3	16
Turdus merula	Blackbird		27	10	11	48
Turdus philomelos	Song thrush	Forest	8	0	0	8
Turdus viscivorus	Mistle thursh	Forest	0	1	0	1
			501	456	569	1,526
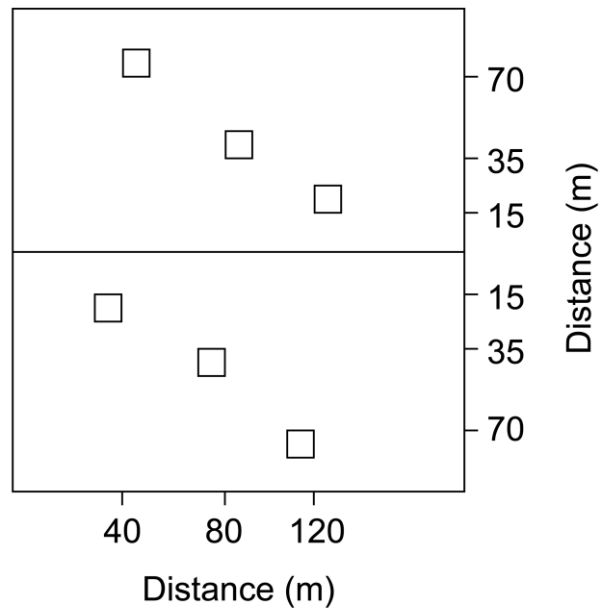
Figure A1.1. Distribution of the 6 plots of habitat sampling within a station. The central line represents the pathway followed during the transect census.
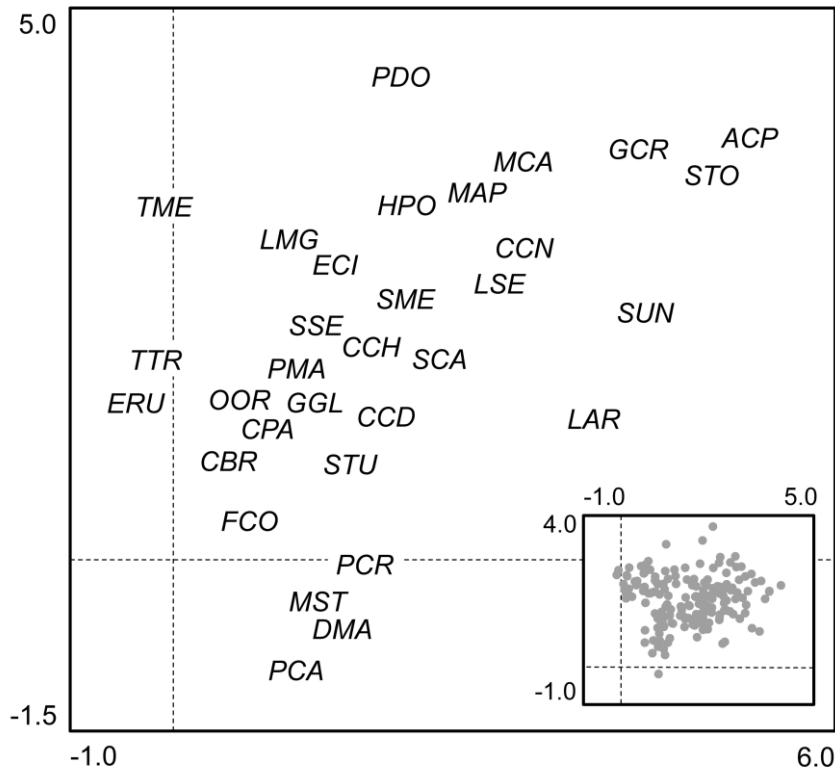
Figure A1.2. Plot of the first two partial DCA axes, showing each species position. The small plot shows the samples distribution in the same axes (ACP: *Anthus campestris*, CBR: *Certhia brachydactyla*, CCD: *Carduelis carduelis*, CCH: *Carduelis chloris*, CCN: *Carduelis cannabina*, CPA: *Columba palumbus*, DMA: *Dendrocopos major*, ECI: *Emberiza cirlus*, ERU: *Erithacus rubecula*, FCO: *Fringilla coelebs*, GCR: *Galerida cristata*, GGL: *Garrulus glandarius*, HPO: *Hippolais polyglotta*, LAR: *Lullula arborea*, LMG: *Luscinia megarhynchos*, LSE: *Lanius senator*, MAP: *Merops apiaster*, MCA: *Miliaria calandra*, MST: *Muscicapa striata*, PCA: *Parus caeruleus*, PCR: *Parus cristatus*, PDO: *Passer domesticus*, PMA: *Parus major*, SCA: *Sylvia cantillans*, SME: *Sylvia melanocephala*, SSE: *Serinus serinus*, STO: *Saxicola torquata*, STU: *Streptopelia turtur*, SUN: *Sylvia undata*, TME: *Turdus merula*, TTR: *Troglodytes troglodytes*).

Annex 2 (capítol 2)

Table A2.1. Details of the burned sites included in the study, showing their size (in hectareas), total number of samples, and number of samples of each postfire management treatment.

Site	Size	Samples	Postfire management[1]		
			UL	*LS*	*HS*
West Mediterranean Basin					
Balsareny	963	33	2	-	31
Borges del Camp	120	6	6	-	-
Caldes1	86	2	-	2	-
Caldes2	70	4	4	-	-
Cardona	1,438	42	-	7	35
Castellbell	391	4	2	-	2
Castellbisbal	224	8	-	2	6
Cistella	250	3	1	1	1
Maçanet	1,255	20	10	4	6
Masquefa	93	6	6	-	-
Montanissell	82	5	4	1	-
Ogern	232	2	-	-	2
Pobla de Massaluca	105	8	8	-	-
Platja d'Aro	444	18	18	-	-
Rocafort	839	34	16	18	-
Riba-roja	642	35	35	-	-
Sant Feliu	514	36	36	-	-
Sant Llorenç	4,559	26	2	3	21
Talamanca	191	13	-	-	13
Tivissa	55	2	2	-	-
Vandellós	1,211	15	13	-	2
Ventalló	1,011	22	1	1	20
Vimbodí	126	4	-	-	4
		348	170	39	139
Northern Rocky Mountains					
Ball Creek	3,082	26	13	5	8
Beta Doris	2,118	73	14	39	20
Big Creek	549	7	6	1	-
Blackfoot Complex	5,754	46	13	15	18
Black Mountain	2,861	89	72	5	12
Boles Meadow	1,807	54	37	13	4
Cooney Ridge	9,767	45	45	-	-
Crazy Horse	4,510	57	21	30	6
Howard Creek	1,144	1	1	-	-
Mineral Peak	7,137	60	60	-	-
Robert1	5,548	97	34	50	13
Robert2	15,813	63	63	-	-
Thompson Creek	15,471	50	31	10	9
Trapper Creek	7,567	49	49	-	-
Snow Talon	13,189	13	8	5	-
Wedge1	9,689	100	18	48	34
Wedge2	12,040	59	59	-	-
		889	544	221	124

[1]*UL: unlogged; LS: light salvage logging; HS: heavy salvage logging.*

Table A2.2. Results of GLMM for the five most common species in each postfire treatment in each region, and relative frequency of each species in each postfire treatment category.

		N^1	P	Postfire treatment[2]		
				UL	LS	HS
Western Mediterranean Basin						
HSD	Blue Tit	49	>0.001	0.25	0.13	0.01
	Great Tit	40	0.250	0.14	0.15	0.08
	Crested Tit	37	0.002	0.18	0.10	0.01
	Eurasian Chaffinch	33	0.007	0.16	0.10	0.01
	Short-toed Treecreeper	25	0.021	0.12	0.10	0.01
ISD	European Serin	92	0.694	0.28	0.28	0.24
	European Blackbird	66	0.023	0.24	0.26	0.12
	Common nightingale	47	0.328	0.16	0.18	0.09
	Winter wren	42	0.384	0.14	0.15	0.09
	European Goldfinch	41	0.014	0.18	0.10	0.05
NSD	Sardinian Warbler	171	0.073	0.55	0.51	0.41
	Subalpine Warbler	155	>0.001	0.28	0.38	0.66
	Wood Lark	126	>0.001	0.22	0.51	0.50
	Dartford Warbler	107	0.072	0.24	0.44	0.35
	Common Stonechat	101	>0.001	0.11	0.46	0.46
Northern Rocky Mountains						
HSD	Yellow-rumped Warbler	198	>0.001	0.27	0.17	0.10
	Pine Siskin	197	0.256	0.22	0.26	0.19
	Western Tanager	110	0.251	0.14	0.10	0.11
	Mountain Chickadee	84	0.015	0.12	0.06	0.05
	Brown Creeper	80	0.068	0.10	0.11	0.03
ISD	Dark-eyed Junco	549	0.107	0.64	0.62	0.53
	Chipping Sparrow	231	0.659	0.26	0.27	0.23
	Mountain Bluebird	171	0.272	0.17	0.22	0.22
	American Robin	138	0.039	0.18	0.11	0.13
	Townsend's Solitaire	58	0.522	0.06	0.06	0.09
NSD	McGillivray's Warbler	168	0.510	0.20	0.19	0.15
	Lazuli Bunting	70	0.500	0.09	0.06	0.07
	Warbling Vireo	44	0.011	0.03	0.06	0.10
	Dusky Flycatcher	39	0.027	0.03	0.06	0.08
	Lincoln's Sparrow	27	-	0.04	0.02	0.00

[1]*N: number of samples with presence of the species*
[2]*UL: unlogged; LS: light salvage logging; HS: heavy salvage logging*

Table A2.3. List of all species detected and their scores in each snag-dependence factor.

Common name	Scientific name	N^1	Snag dependence[2]			
			H	N	F	SDV
Western Mediterranean Basin						
Blue Tit	*Parus caeruleus*	49	2	1	1	4
Great Tit	*Parus major*	40	2	1	1	4
Crested Tit	*Parus cristatus*	37	2	1	1	4
Short-toed Treecreeper	*Certhia brachydactyla*	25	2	1	1	4
Firecrest	*Regulus ignicapillus*	14	2	1	1	4
Great Spotted Woodpecker	*Dendrocopos major*	13	2	1	1	4
Eurasian Golden Oriole	*Oriolus oriolus*	7	2	1	1	4
Long-tailed Tit	*Aegithalos caudatus*	6	2	1	1	4
Coal Tit	*Parus ater*	1	2	1	1	4
Eurasian Chaffinch	*Fringilla coelebs*	33	2	1	0	3
Wood Pigeon	*Columba palumbus*	22	2	1	0	3
European Jay	*Garrulus glandarius*	21	2	1	0	3
Bonelli's Warbler	*Phylloscopus bonelli*	16	2	0	1	3
Common Cuckoo	*Cuculus canorus*	3	2	0	1	3
Song Thrush	*Turdus philomelos*	2	2	1	0	3
European Blackbird	*Turdus merula*	66	2	0	0	2
Blackcap	*Sylvia atricapilla*	15	2	0	0	2
Spotted Flycatcher	*Muscicapa striata*	4	2	0	0	2
Cetti's Warbler	*Cettia cettia*	2	2	0	0	2
Eurasian Green Woodpecker	*Picus viridis*	2	1	1	0	2
Stock Dove	*Columba oenas*	1	1	1	0	2
Common Chiffchaff	*Phylloscopus collybita*	1	2	0	0	2
European Serin	*Serinus serinus*	92	1	0	0	1
Common Nightingale	*Luscinia megarhynchos*	47	1	0	0	1
Winter Wren	*Troglodytes troglodytes*	42	1	0	0	1
European Goldfinch	*Carduelis carduelis*	41	1	0	0	1
European Greenfinch	*Carduelis chloris*	19	1	0	0	1
Cirl Bunting	*Emberiza cirlus*	18	1	0	0	1
European Robin	*Erithacus rubecula*	18	1	0	0	1
European Starling	*Sturnus vulgaris*	10	0	1	0	1
European Turtle-dove	*Streptopelia turtur*	9	1	0	0	1
Black-billed Magpie	*Pica pica*	4	0	1	0	1
Common Buzzard	*Buteo buteo*	3	0	1	0	1
Short-toed Snake-eagle	*Circaetus gallicus*	2		1	0	1
Eurasian Nightjar	*Caprimulgus europaeus*	1	1	0	0	1
Carrion Crow	*Corvus corone*	1	0	1	0	1
Eurasian Hobby	*Falco subbuteo*	1		1	0	1
Eurasian Hoopoe	*Upupa epops*	1	1	0	0	1
Sardinian Warbler	*Sylvia melanocephala*	171	0	0	0	0
Subalpine Warbler	*Sylvia cantillans*	155	0	0	0	0
Wood Lark	*Lullula arborea*	126	0	0	0	0
Dartford Warbler	*Sylvia undata*	107	0	0	0	0
Common Stonechat	*Saxicola torquata*	101	0	0	0	0
Melodious Warbler	*Hippolais polyglotta*	89	0	0	0	0
Eurasian Linnet	*Carduelis cannabina*	71	0	0	0	0
Rock Bunting	*Emberiza cia*	65	0	0	0	0

Common name	Scientific name	N[1]	Snag dependence[2]			
			H	N	F	SDV
Corn Bunting	*Emberiza calandra*	62	0	0	0	0
Black-eared Wheatear	*Oenanthe hispanica*	58	0	0	0	0
Woodchat Shrike	*Lanius senator*	34	0	0	0	0
House Sparrow	*Passer domesticus*	29	0	0	0	0
Thekla Lark	*Galerida theklae*	22	0	0	0	0
Tawny Pipit	*Anthus campestris*	20	0	0	0	0
Ortolan Bunting	*Emberiza hortulana*	20	0	0	0	0
European Bee-eater	*Merops apiaster*	17	0	0	0	0
Red-legged Partridge	*Alectoris rufa*	14	0	0	0	0
Rock Sparrow	*Petronia petronia*	14	0	0	0	0
Crested Lark	*Galerida cristata*	13	0	0	0	0
Southern Grey Shrike	*Lanius meridionalis*	8	0	0	0	0
Blue Rock-thrush	*Monticola solitarius*	8	0	0	0	0
Zitting Cisticola	*Cisticola juncidis*	5	0	0	0	0
Common Kestrel	*Falco tinnunculus*	5	0	0	0	0
White Wagtail	*Motacilla alba*	5	0	0	0	0
Black Wheatear	*Oenanthe leucura*	5	0	0	0	0
Black Redstart	*Phoenicurus ochruros*	5	0	0	0	0
Rufous-tailed Rock-thrush	*Monticola saxatilis*	4	0	0	0	0
Little Owl	*Athene noctua*	2	0	0	0	0
Red-backed Shrike	*Lanius collurio*	1	0	0	0	0
Tree Sparrow	*Passer montanus*	1	0	0	0	0
Common Pheasant	*Phasianus colchicus*	1	0	0	0	0
Orphean Warbler	*Sylvia hortensis*	1	0	0	0	0
Northern Rocky Mountains						
Yellow-rumped Warbler	*Dendroica coronata*	292	2	1	1	4
Western Tanager	*Piranga ludoviciana*	145	2	1	1	4
Mountain Chickadee	*Poecile gambeli*	117	2	1	1	4
Red-breasted Nuthatch	*Sitta canadensis*	108	2	1	1	4
Brown Creeper	*Certhia americana*	106	2	1	1	4
Hairy Woodpecker	*Picoides villosus*	99	2	1	1	4
Three-toed Woodpecker	*Picoides tridactylus*	52	2	1	1	4
Black-backed Woodpecker	*Picoides arcticus*	50	2	1	1	4
Ruby-crowned Kinglet	*Regulus calendula*	42	2	1	1	4
Red Crossbill	*Loxia curvirostra*	41	2	1	1	4
Cassin's Vireo	*Vireo cassinii*	17	2	1	1	4
Townsend's Warbler	*Dendroica townsendi*	15	2	1	1	4
Red-naped Sapsucker	*Sphyrapicus nuchalis*	10	2	1	1	4
Chestnut-backed Chickadee	*Poecile rufescens*	4	2	1	1	4
Pileated Woodpecker	*Dryocopus pileatus*	2	2	1	1	4
Williamson's Sapsucker	*Sphyrapicus thyroideus*	2	2	1	1	4
Pine Siskin	*Carduelis pinus*	294	2	1	0	3
Hammond's Flycatcher	*Empidonax hammondi*	82	2	1	0	3
Black-capped Chickadee	*Poecile atricapilla*	33	1	1	1	3

151

Common name	Scientific name	N[1]	Snag dependence[2]			
			H	N	F	SDV
Cassin's Finch	*Carpodacus cassinii*	28	2	1	0	3
Gray Jay	*Perisoreus canadensis*	21	2	1	0	3
Steller's Jay	*Cyanocitta stelleri*	11	2	1	0	3
Golden-crowned Kinglet	*Regulus satrapa*	8	2	1	0	3
Downy Woodpecker	*Picoides pubescens*	4	1	1	1	3
White-breasted Nuthatch	*Sitta carolinensis*	4	1	1	1	3
Cooper's Hawk	*Accipiter cooperi*	3	2	1	0	3
Clark's Nutcracker	*Nucifraga columbiana*	2	2	1	0	3
Evening Grosbeak	*Coccothraustes vespertinus*	1	2	1	0	3
Dark-eyed Junco	*Junco hyemalis*	832	2	0	0	2
Chipping Sparrow	*Spizella passerina*	368	1	1	0	2
Mountain Bluebird	*Sialia currucoides*	237	1	1	0	2
Swainson's Thrush	*Catharus ustulatus*	79	2	0	0	2
Northern Flicker	*Colaptes auratus*	68	1	1	0	2
Olive-sided Flycatcher	*Contopus cooperi*	32	1	1	0	2
Western Wood-Pewee	*Contopus sordidulus*	22	1	1	0	2
Hermit Thrush	*Catharus guttatus*	12	2	0	0	2
Winter Wren	*Troglodytes troglodytes*	11	0	2	0	2
Cedar Waxwing	*Bombycilla cedrorum*	10	1	1	0	2
Pine Grosbeak	*Pinicola enucleator*	7	2	0	0	2
Western Bluebird	*Sialia mexicana*	6	1	1	0	2
Red-tailed Hawk	*Buteo jamaicensis*	5	1	1	0	2
American Kestrel	*Falco sparverius*	4	1	1	0	2
Mourning Dove	*Zenaida macroura*	3	1	1	0	2
Bullock's Oriole	*Icterus bullocki*	1	1	1	0	2
Great Horned Owl	*Bubo virginianus*	1	1	1	0	2
Lewis' Woodpecker	*Melanerpes lewis*	1	1	1	0	2
Northern Hawk Owl	*Surnia ulula*	1	1	1	0	2
American Robin	*Turdus migratorius*	245	0	1	0	1
Townsend's Solitaire	*Myadestes townsendi*	82	1	0	0	1
Brown-headed Cowbird	*Molothrus ater*	42	0	1	0	1
House Wren	*Troglodytes aedon*	36	0	1	0	1
Wilson's Warbler	*Wilsonia pusilla*	30	1	0	0	1
Song Sparrow	*Melospiza melodia*	28	1	0	0	1
Calliope Hummingbird	*Stellula calliope*	23	1	0	0	1
Northern Waterthrush	*Seiurus noveboracensis*	17	1	0	0	1
Common Yellowthroat	*Geothlyphis trichas*	8	1	0	0	1
Ruffed Grouse	*Bonasa umbellus*	5	1	0	0	1
American Goldfinch	*Carduelis tristis*	2	1	0	0	1
Blue Grouse	*Dendragapus obscurus*	2	1	0	0	1
American Crow	*Corvus brachyrynchos*	1	0	1	0	1
Common Nighthawk	*Chordeiles minor*	1	1	0	0	1
Eastern Kingbird	*Tyrannus tyrannus*	1	1	0	0	1
European Starling	*Sturnus vulgaris*	1	0	1	0	1

Common name	Scientific name	N^1	Snag dependence[2]			
			H	*N*	*F*	*SDV*
MacGillivray's Warbler	*Oporornis tolmiei*	228	0	0	0	0
Lazuli Bunting	*Passerina amoena*	88	0	0	0	0
Warbling Vireo	*Vireo gilvus*	62	0	0	0	0
Dusky Flycatcher	*Empidonax oberholseri*	53	0	0	0	0
Lincoln's Sparrow	*Melospiza lincolnii*	53	0	0	0	0
White-crowned Sparrow	*Zonotrichia leucophrys*	33	0	0	0	0
Black-headed Grosbeak	*Pheucticus melanocephalus*	18	0	0	0	0
Fox Sparrow	*Passerella iliaca*	17	0	0	0	0
Orange-crowned Warbler	*Vermivora celata*	17	0	0	0	0
Rufous Hummingbird	*Selasphorus rufus*	14	0	0	0	0
Willow Flycatcher	*Empidonax traillii*	10	0	0	0	0
Rock Wren	*Salpinctes obsoletus*	7	0	0	0	0
Yellow Warbler	*Dendroica petechia*	4	0	0	0	0
Nashville Warbler	*Vermivora ruficapilla*	3	0	0	0	0
Gray Catbird	*Dumetella carolinensis*	1	0	0	0	0
Horned Lark	*Eremophila alpestris*	1	0	0	0	0
Spotted Towhee	*Pipilo maculatus*	1	0	0	0	0

[1]*N: number of samples with presence of the species*
[2]*H: Habitat requirements, N: nest location, F: foraging behavior, SDV: snag-dependence value*

Annex 3

Figura A3.1. En primer terme, pilons de fusta de pi blanc triturada per a combustible de biomassa. Al darrere es veuen els pins tallats, tallats esperant a ser triturats. Incendi de Ventalló, juny de 2009.

Figura A3.2. Regeneració de plantes sota un piló de branques cremades construït a l'incendi de Sant Llorenç, algunes de les quals fan fruits carnosos i són dispersades per ocells (esparraguera, llentiscle, romaguera, rogeta). Maig de 2009.

Figura A3.3. Rodal tallat al foc de Black Mountain, a Montana, juny de 2010 (gairebé set anys després del foc). Al fons es veu la vall de Missoula i més enllà, les Mission Mountains nevades.

Figura A3.4. Zona tallada al foc de Ventalló, febrer de 2010. Al vessant de la dreta es veuen algunes terrasses, prova que la pineda cremada havia estat una zona treballada anteriorment, possiblement de conreus d'oliveres. Al fons s'endevinen les muntanyes de Serra Verdera. La regeneració de la vegetació (tres anys i mig després del foc) contrasta amb la de la foto anterior de Montana.

Figura A3.5. Regeneració intensa de pi blanc en una zona tallada dins de l'incendi de Ventalló, febrer de 2010 (tres anys i mig després del foc).

Figura A3.6. Vista del Montgrí en un dia encalitjat des del puig Segalar, el punt més alt de la zona cremada de Ventalló. Juliol de 2007 (gairebé un any després del foc).
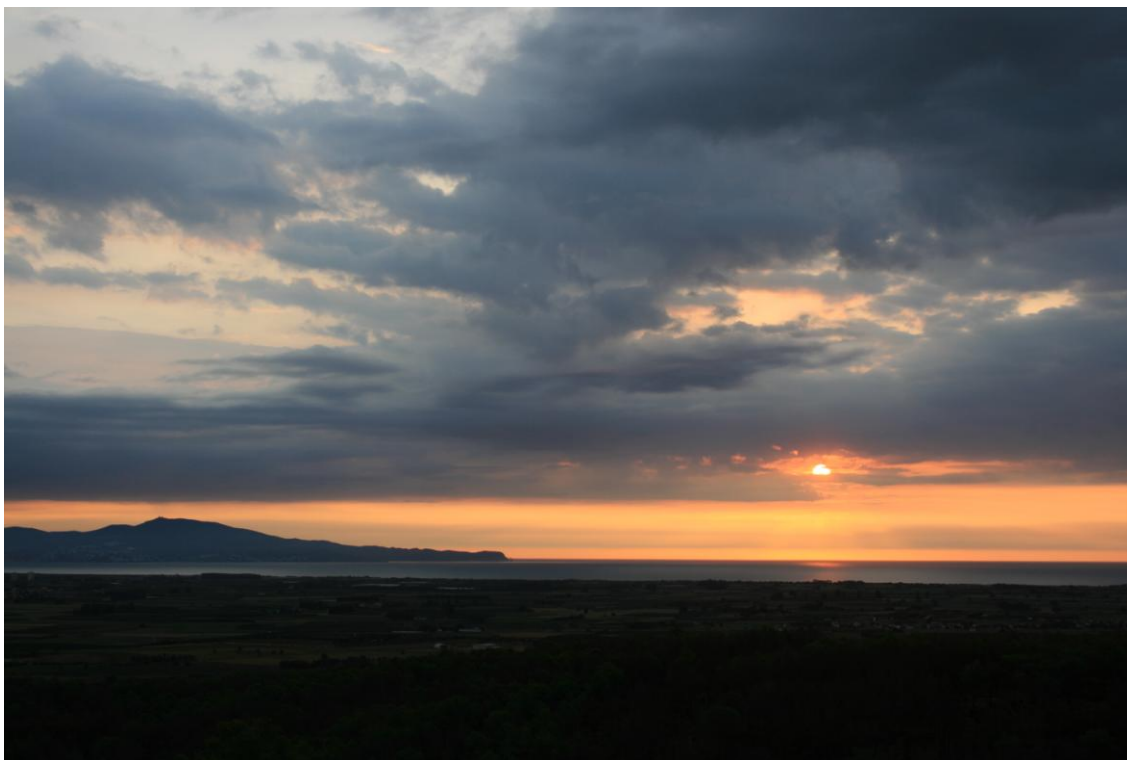
Figura A3.7. I per acabar, una foto de les vistes privilegiades que tenia des del meu lloc de treball durant la tesi!