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Fragmentation and human disturbances in peri-urban forests: effects on vascular flora

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per optar al grau de Doctora

Amb el vist-i-plau dels directors de tesi:

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Sumari

Introducció general

CONCEPTES BÀSICS

El paisatge i els seus elements

El terme paisatge ha admès definicions molt diferents al llarg de la història depenent des de la disciplina des de la qual s'ha estudiat (Pino & Rodà 1999). Tractaments més o menys globals del paisatge s'han abordat des de diferents àmbits del coneixement, i especialment des de la geografia (Nogué 1984; Bolòs 1992). Dins de la ciència de la conservació, la disciplina que ha desenvolupat el marc conceptual per a entendre els processos que tenen lloc a escala de paisatge és l'anomenada 'ecologia del paisatge', la qual considera aquest com una unitat funcional més o menys repetible al llarg d'un determinat territori i integrada per un conjunt d'ecosistemes interrelacionats (Forman & Gordon 1986; Forman, 1995a). A més, l'ecologia del paisatge inclou l'home com a agent modulador del territori. Per aquest motiu, en aquells territoris fortament antropitzats, el paisatge es considera el resultat de la interacció entre natura i cultura (Zonneveld 1995; Farina 1998).

Cada paisatge està format per un conjunt d'unitats que, de forma genèrica, han estat anomenades elements del paisatge (*landscape elements*, Forman & Godron, 1986). La unitat espacial mínima dins un determinat paisatge s'anomena unitat de paisatge (*landscape unit*). Forman & Godron (1986) proposen un model de descripció del paisatge en el qual es consideren tres elements que corresponen a les unitats de paisatge: les tessel·les (*patches*), els corredors (*corridors*) i la matriu subjacent (*matrix*). **Les tessel·les** són superfícies amb característiques relativament homogènies i diferents de les del seu voltant, les quals poden variar en la seva mida, forma, heterogeneïtat, tipus de vora, etc. **Els corredors** són definits com a elements (tessel·les) lineals del paisatge (com per exemple camins, tanques verdes o rius). Finalment, es considera que **la matriu** és el tipus de tessel·la que ocupa major superfície en el paisatge o, en el seu defecte, l'element amb un grau d'interconnexió més elevat o amb més influència en la dinàmica del paisatge.

Les unitats del paisatge poden patir canvis al llarg de l'espai i el temps, i aquests canvis constitueixen la dinàmica del paisatge (Forman & Gordon 1986). Per exemple, una clapa de bosc pot créixer al llarg dels anys o bé trencar-se en moltes unitats més petites. A la vegada, poden aparèixer nous camps de cultiu i noves cases entre aquests fragments de bosc.

Dinàmiques del paisatge: la fragmentació

Una de les dinàmiques paisatgístiques que ha estat més estudiada és la fragmentació, ja que es considera una forma específica de degradació dels hàbitats originada tant per causes naturals com antròpiques (Forman 1995a; Haila 2002). La fragmentació s'ha definit de forma general com un procés que actua a escala de paisatge i que inclou la pèrdua d'hàbitat i el seu trencament en diverses unitats (Forman 1995a; Saunders et al. 1991; Meffe 1994; Fahrig 2003; Figura 1). La fragmentació i la pèrdua d'hàbitats naturals associada constitueixen un component essencial del canvi global (Vitousek 1994).

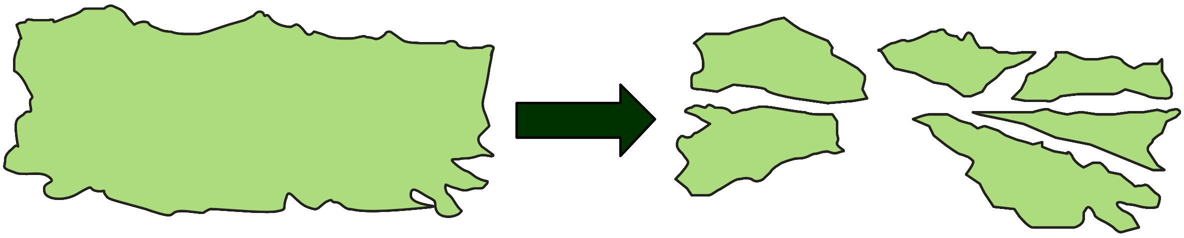

Figura 1. Fragmentació d'un element paisatgístic.

No obstant, cal remarcar que la fragmentació és tan sols un dels processos que intervenen en el procés de transformació del paisatge. Forman (1995a) identifica tot un seguit de processos espacials que intervenen en el procés transformador del territori, i que sovint actuen conjuntament: **la perforació d'un hàbitat** (e.g., l'aparició de clarianes dins d'un bosc); **la dissecció** (el trencament de l'hàbitat que origina el pas d'una carretera o una via de tren); **la fragmentació** (trencament de l'habitat en unitats més petites); **la reducció** (la disminució de l'àrea d'un hàbitat sense que hi hagi trencament); **i la desaparició** de l'hàbitat. Per tant, el procés de fragmentació s'ha de considerar conjuntament amb els altres processos espacials.

En una revisió recent dels efectes de la fragmentació dels hàbitats sobre la biodiveristat, Fahrig (2003) suggereix que el procés de fragmentació, en sentit ample, inclou dos components: la pèrdua d'hàbitat total i la fragmentació "*per se*", és a dir els canvis en la configuració d'aquest hàbitat que resulten del trencament de l'hàbitat original (l'augment en el nombre de clapes, la disminució de la mida de les clapes i l'augment en l'aïllament entre clapes). Fahrig (2003) proposa en el seu estudi que el terme "fragmentació" s'ha de reservar per aquelles situacions en les quals hi ha un trencament de l'hàbitat, independentment de si hi ha pèrdua o no d'aquest. No obstant, si s'analitza la literatura existent sobre la fragmentació dels hàbitats, s'observa que la interpretació varia molt entre estudis, ja que pot incloure la totalitat dels efectes esmentats o tan sols una part.

D'altra banda, es tendeix a assimilar totes les taques d'hàbitat d'un territori a fragments, és a dir, a la resultant del procés de fragmentació. Això no te perquè ser cert, ja que algunes taques es poden originar de nou per canvi d'usos (vegeu, per exemple, els capítols 1, 2, 3, i 4 d'aquest treball). Per tal d'evitar aquesta confusió entre el procés de fragmentació i la pauta que en resulta, hem optat per utilitzar els termes "clapa" o "taca" en comptes de "fragment" de bosc. A més, sovint hem emprat el terme "fragmentació" de forma genèrica per a referir-nos al conjunt de processos que, en una escala temporal de segles, han comportat la transformació de paisatges inicialment forestals en mosaics agroforestals i peri-urbans.

Efectes de la fragmentació i les seves conseqüències

La fragmentació entesa com el trencament, reducció i aïllament dels hàbitats té conseqüències evidents sobre la mida, la forma, el nombre, i el grau d'aïllament dels fragments resultants. Aquests efectes tenen, a la vegada, importants conseqüències sobre les espècies que integren els hàbitats (Soulé 1987; Forman 1995a), i també sobre molts dels seus processos ecològics (Pickett & White 1985; Forman 1995a).

Molts estudis han demostrat que la fragmentació té efectes importants sobre la riquesa i diversitat específica d'un determinat hàbitat (e.g., Forman et al. 1976; Forman 1995b; Tellería 1997). S'ha observat que, en general, el nombre d'espècies és sovint funció creixent de la superfície del fragment. Una disminució de la seva mida determina, en general, una disminució de les poblacions de molts organismes (e.g., Forman 1995a; Tellería 1997). Si la mida del fragment és inferior a la mínima viable per a una determinada espècie, aquesta desapareixerà (Meffé & Carroll 1994). No obstant, un cert nombre d'individus es pot mantenir si hi ha processos d'immigració continus des d'una "àrea font" (Pulliam 1988; Eriksson 1996).

A la vegada, la fragmentació també implica un augment del perímetre de contacte entre els fragments d'hàbitat i els usos del sòl adjacents. A més, la relació perímetre/àrea de cada fragment creix, augmentant per tant l'anomenat "efecte marge" (Forman & Gordon 1986). Els canvis en la forma del fragment també tenen importants conseqüències en l'efecte marge, ja que en formes més convolutes augmentarà el perímetre de contacte (Forman 1995b; Ries et al. 2004). A grans trets, l'efecte marge identifica un gradient de condicions ambientals des de la vora fins a l'interior del fragment: al marge del fragment (també anomentat ecotó), sovint hi trobem una elevada diversitat biològica degut a l'arribada d'espècies de l'hàbitat adjacent o a la presència d'espècies multi-hàbitat, mentre que a l'interior, la diversitat disminueix i hi predominen les espècies més específiques de l'hàbitat del fragment (Forman & Gordon 1986; Murcia 1995; Cadenasso & Pickett 2001; Ries et al. 2004). Diversos estudis han demostrat que les espècies pròpies d'interior es veuen afectades negativament per la fragmentació, ja que passen a estar més exposades a pertorbacions i a l'arribada d'espècies dels hàbitats adjacents (Forman & Gordon 1986; Luken 1997). Per tant, atès que hi ha una banda perimetral on es manifesta l'efecte marge, l'establiment d'unes condicions típiques d'interior dependrà de la mida de la tessel·la, però també de la importància relativa de la zona interior i del marge.

Un altre efecte de la fragmentació és l'aïllament dels fragments, el qual també determina la capacitat de les espècies a propagar-se d'un fragment a l'altre. Per exemple, als fragments de bosc més allunyats d'altres boscos els serà més difícil d'ésser colonitzats per noves espècies que no pas fragments més propers. No obstant, els efectes de la fragmentació varien molt entre espècies, ja que depenen entre d'altres de la capacitat de dispersió d'aquelles, del context ecològic o de la història prèvia (Murcia 1995, Ries et al. 2004).

ANTECEDENTS EN L'ESTUDI DE LA FRAGMENTACIÓ

La fragmentació dels hàbitats ha estat àmpliament estudiada en el marc de la biologia de la conservació i l'ecologia del paisatge (e.g., Forman 1995a,b; Debinski & Holt 2000; Haila 2002). Des dels seus orígens, els estudis sobre fragmentació i degradació dels hàbitats han estat inspirats en la teoria de la biogeografia insular (Mc Arthur & Wilson 1967), la qual presenta les illes oceàniques com a ambients estables i, sota aquesta estabilitat, prediu el nombre d'espècies en equilibri, el qual depèn de les taxes d'immigració i extinció, de la mida de les illes (efecte e l'àrea *per se*) i de la distància entre illes o entre illa i continent. També els fragments d'hàbitat (o les clapes) es poden considerar "illes" envoltades d'hàbitats diferents comparables al mar, sovint originades per l'acció de l'home (Haila 2002). Corol·laris posteriors a aquesta teoria i referents a la relació espècies-àrea també han influït molt en els estudis sobre fragmentació, com la hipòtesis de diversitat d'hàbitats (Connor & McCoy 1979; Coleman 1981), que proposa que els fragments de major mida tenen una diversitat superior d'hàbitats (i per tant més variació ambiental) i això determina un increment en el seu nombre d'espècies. Cal remarcar, però, que els fragments d'hàbitat rarament són un sistema en equilibri (Forman & Gordon 1986). Per aquest motiu, en la última dècada diversos autors han posat en dubte l'aplicabilitat de la teoria d'illes en fragments continentals, ja que no té en compte molts factors com el dinamisme dels ecosistemes (canvis en el temps i en l'espai), les variacions ambientals dins dels fragments, l'efecte marge (és a dir, el gradient de condicions ambientals que va des de la vora fins al centre del fragment (Forman & Gordon 1986), l'aport de nutrients, etc. (e.g., Saunders *et al*. 1991; Haila 1990; Anderson *et al*. 2001). Per tant, malgrat que la relació espècies-àrea és una de les generalitzacions més robustes en ecologia (Holt et al. 1999; Fangliang & Legendre 2002), les causes que expliquen aquesta relació són molt diverses i difícils de determinar (Schoereder *et al.* 2004). Entre elles cal esmentar els possibles artefactes derivats del mostreig (Preston 1962): en un mostreig a l'atzar, el nombre de mostres augmenta amb la superfície de la taca d'hàbitat, i això sol ja incrementa les probabilitats de trobar-hi espècies. A banda d'aquest possible efecte de mostreig, el nombre d'espècies per clapa pot augmentar amb la mida de la clapa bàsicament per dos motius: pel que s'ha nomenat efectes de l'àrea *per se* i per la major heterogeneïtat ambiental esperable en clapes grans. Segons la hipòtesi de l'àrea *per se,* s'espera un augment del nombre d'espècies per clapa amb la superfície total de la taca, a causa de processos biològics diversos com els que regulen les taxes d'immigració i d'extinció de les espècies. Quan baixem a nivell d'unitat de mostreig (parcel·les d'àrea fixa), aquesta major riquesa específica en clapes grans es tradueix també en un major número d'espècies per unitat de mostreig, suposant que dos determinants clau de la relació espècies-àrea (el rang d'abundància de les espècies, i la distribució espacial dels individus de cada població uniespecífica; He & Legendre 2002) no variïn sistemàticament amb la mida de la clapa. Per altra banda, aquesta aproximació basada en la riquesa de les unitats de mostreig (i no en la riquesa de les clapes) pot servir de base per a discriminar els efectes de l'àrea *per se* d'aquells deguts a variacions ambientals o d'altra mena (Kelly et al. 1989; Benayas et al. 1999). Aquest tipus d'estudis són especial objecte de consideració al present treball.

LA FRAGMENTACIÓ FORESTAL: ELS BOSCOS MEDITERRANIS

La fragmentació del bosc té, en general, efectes negatius sobre el nombre i composició de les espècies de plantes presents; a la vegada, també determina propietats fonamentals d'aquests fragments com és ara la seva relació perímetre-àrea (Forman & Godron 1986), que es pot relacionar directament amb el grau d'influència dels hàbitats adjacents. En general, s'ha observat que els efectes de la fragmentació forestal són negatius per a la riquesa i composició específica, ja que es produeix una disminució de les espècies més sensibles per efectes directes o indirectes de l'activitat antròpica (Holt et al. 1995; Hanski

1999; Bascompte et al. 2001; Hill 2001). No obstant, la fragmentació també pot afavorir l'entrada d'espècies no forestals, principalment en els seus marges (e.g., Hobbs 1988, 2000; Wiens 1992; Murcia 1995). Aquest darrer procés pot ser especialment acusat en àrees molt pertorbades per l'home, on la riquesa i composició específica podran venir més determinades per les pertorbacions que es produeixen dins i a l'entorn del bosc que no pas per les característiques intrínseques d'aquest (mida, forma, grau d'aïllament, estructura forestal).

Com a poques altres regions del món, els humans han modelat els paisatges mediterranis durant centenars d'anys a través de pràctiques com els incendis, tales, conreus, o la construcció de vivendes (Trabaud 1981; Barbéro et al. 1990; Blondel & Aronson 1999). Els progressius canvis d'usos del sòl han causat la fragmentació dels boscos mediterranis, principalment en aquelles zones àmpliament habitades, a l'inici com a conseqüència de l'activitat agrícola i posteriorment per processos d'industrialització, urbanització i la construcció infrastructures (Debussche et al. 1999; Herrando 2001). Durant el segle XX, els boscos mediterranis han patit tot un seguit de processos contrastats pel que fa a la seva estructura i extensió: a les zones de muntanya, l'abandonament dels camps de cultiu i les pastures ha implicat l'extensió del bosc i de comunitats arbustives (Preiss et al. 1997). En canvi, a les valls fèrtils i a la costa l'activitat humana s'ha intensificat com a resultat de l'agricultura i les urbanitzacions, i els fragments de vegetació natural han esdevingut cada cop més escassos i aïllats (Debussche et al. 1999; Herrando 2001). Aquest procés de transformació dels paisatges agroforestals pot tenir efectes negatius per a la conservació dels ecosistemes i de les espècies pròpies d'aquests ambients (Preiss et al. 1997).

Característiques diferencials dels boscos peri-urbans: la influència antròpica

La reducció de l'àrea forestal i l'efecte dels hàbitats adjacents sobre la riquesa i composició específica dels boscos mediterranis esdevé especialment evident en zones urbanes i peri-urbanes on, a les transformacions històriques del paisatge, se li sumen les pertorbacions antròpiques actuals, característiques d'un territori densament habitat i industrialitzat (Anguera et al. 1994; Ribas 2000; Gonard et al. 2001).

L'extinció d'espècies molt vulnerables i la invasió d'espècies no forestals sembla ésser especialment intensa en boscos fragmentats urbans i peri-urbans (Honnay et al. 1999c; Hobbs 1988,2000; Godefroid & Koedam 2003b). No obstant, l'efecte de les àrees peri-urbanes en la composició i riquesa d'espècies forestals ha rebut una atenció molt escassa fins fa poc (Honnay et al. 1999c), malgrat el fet que aquests boscos estan fortament influenciats per l'activitat antròpica (e.g., Matlack 1993; Forman & Alexander 1998; Godefroid & Koedam 2003b).

En el cas dels boscos mediterranis, la major part d'estudis s'han centrat en àrees rurals tenint en compte principalment factors relatius a l'estructura del paisatge o a pertorbacions ambientals (e.g., Telleria & Santos 1997,1999; Bascompte & Rodríguez 2001; Brotons & Herrando 2001). A més, la majoria d'aquests es centren en l'estudi d'ocells o d'altres vertebrats. En canvi, l'efecte de l'activitat humana pel que fa a plantes vasculars ha estat poc considerada (e.g., Gallego et al. 2004), i encara menys en entorns peri-urbans. La freqüentació antròpica deguda als usos de lleure i l'abandonament de deixalles també determinen l'estat del sotabosc i les espècies de plantes que s'hi poden trobar, i afavoreix l'entrada d'espècies ruderals, les quals poden tenir efectes negatius sobre les espècies de bosc. A la vegada, també apareixen altres espècies lligades a l'activitat humana com ara plantes cultivades provinents de conreus o de jardins. L'activitat humana és el principal causant de la introducció d'espècies al·lòctones arreu del planeta, sobretot en aquelles zones influenciades per l'home com urbanitzacions, ciutats, marges de carreteres, etc. (Sykora 1990; Quézel et al. 1990; Vilà et al. 2001).

La recerca ecològica en aquest context tant complex és molt escassa, i encara se sap ben poc sobre la resposta dels diferents organismes als règims de pertorbació antròpics i als canvis estructurals del paisatge. Aquest tipus d'estudis esdevé, per tant, un repte pels investigadors, que no només han d'intentar entendre el paper de les pertorbacions i de l'estructura dels boscos i del paisatge sobre les espècies sinó també identificar pautes que serveixin per a fer compatibles la gestió del territori i la seva conservació biològica, especialment en aquelles àrees més pertorbades per l'home.

L'ÀREA DEL VALLÈS

El present estudi es desenvolupa a la plana del Vallès, on la massa forestal s'ha vist afectada per tot un seguit de canvis espacials que s'han anat solapant un rere l'altre al llarg de la història, i que han provocat que l'antiga matriu agrícola i forestal tingués cada vegada més enclavaments industrials i urbans. Actualment, la plana vallesana està representada bàsicament per un paisatge agroforestal fortament fragmentat amb una gran quantitat d'aquests enclavaments, fet pel qual s'ha cregut una àrea molt adequada per estudiar els processos de fragmentació i d'alteració dels boscos (Figura 2).

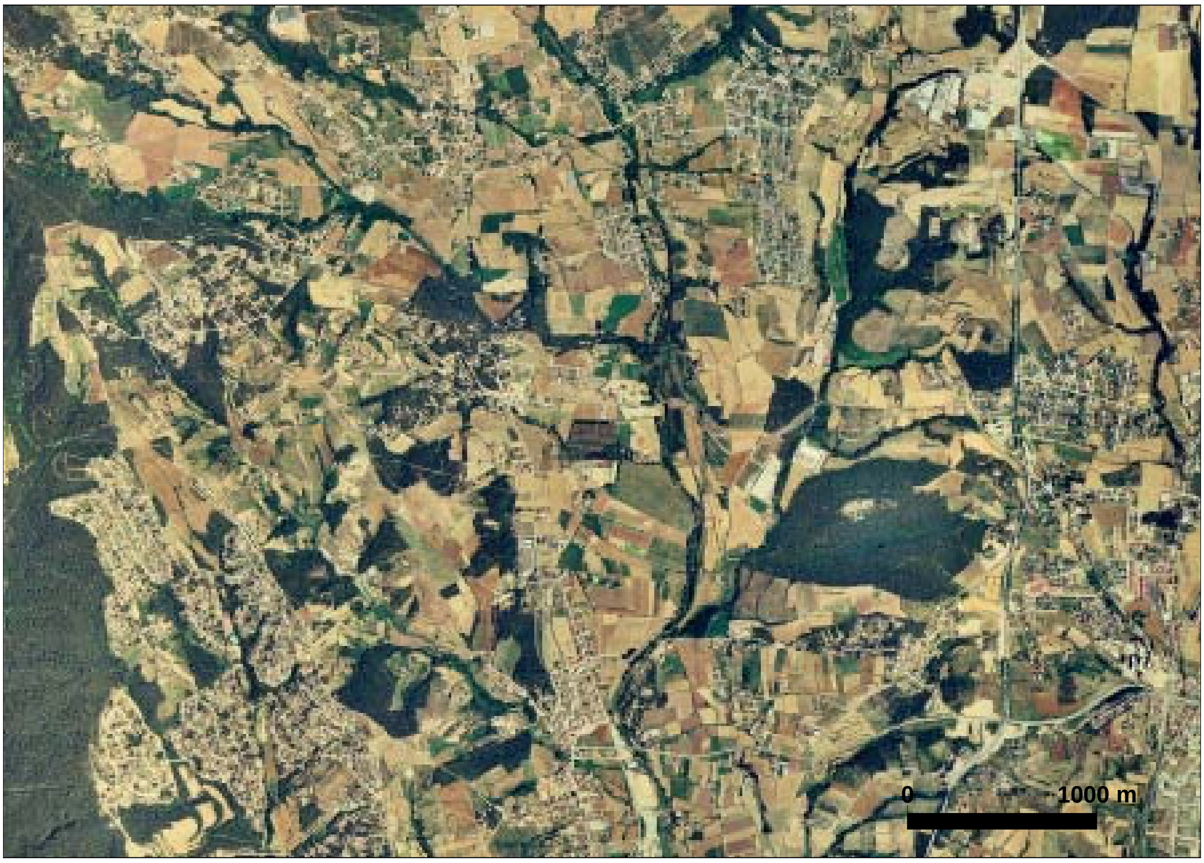

Figura 2. Fotografia aèria d'un fragment de la zona d'estudi (terme municipal de Granollers) on s'observa la matriu agroforestal amb enclavaments urbans i la xarxa viària. Font: ortofotomapa 1:25:000 de Granollers (ICC, 2000).

Durant els segles XVIII i XIX, la regressió de la superfície forestal a la plana vallesana va ser força important, i estava lligada al creixement de la població, a la demanda de productes forestals i als espais de cultiu. L'arribada del ferrocarril també va ser important, ja que va representar un major lligam de la comarca amb la ciutat de Barcelona (Urtega 1987); posteriorment, durant el període de la industrialització, hi va haver importants moviments de població de la muntanya cap a la plana, i la indústria va créixer molt (Boada 1998). Entre 1950 i 1975 es produeix una immigració massiva i la indústria experimenta el seu major creixement; apareixen les infrastructures de transport associades i a la vegada aquests canvis impulsen el fenomen de la construcció, amb la conseqüent aparició de nous enclavaments urbans (Diego et al. 1994). Actualment, el desenvolupament urbanístic al Vallès continua, propiciat sobretot per la proximitat de la plana a la ciutat de Barcelona.

La pressió humana sobre els ecosistemes forestals vallesans s'ha deixat sentir de ben segur amb forta intensitat. La gestió del bosc (tales, podes i estassades principalment), s'ha produït durant centenars d'anys, sobretot quan la fusta, la llenya i el carbó vegetal eren recursos imprescindibles (Boada et al. 1984; Anguera et al. 1994). La freqüentació dels boscos com a àrees de lleure ha estat posterior, arrel del creixement urbà i de la consolidació d'un model de societat bàsicament industrial i de serveis. Aquesta incidència antròpica ha provocat molt probablement canvis en la composició d'espècies, amb un augment de les espècies banals o exòtiques i la disminució de les espècies més sensibles.

Objectius generals

El principal objectiu d'aquesta tesi és analitzar com la fragmentació del bosc i les pertorbacions humanes associades afecten l'estructura, la composició i l'estat de conservació dels boscos en un paisatge peri-urbà mediterrani, tenint en compte factors climàtics, topogràfics, de pertorbació humana i paisatgístics que operen a diverses escales. També s'ha fet especial èmfasi, sempre i quan ha estat possible, en l'interès dels resultats obtinguts pel que fa a la conservació i la gestió de la biodiversitat forestal en aquestes àrees fortament humanitzades.

Aquests objectius generals es concreten en els següents capítols:

Capítol 1: Quercus *and* Pinus *cover are determined by landscape structure and dynamics in peri-urban Mediterranean forests.* Estudiar la importància de les variables ambientals, de les pertorbacions antròpiques i de l'estructura de la clapa i del paisatge sobre el recobriment arbori de *Quercus* i *Pinus*, els quals s'han considerat com a indicadors indirectes de l'estat successional del bosc.

Capítol 2: *Sensitivity of understory species to physical metrics, man-induced disturbance and landscape structure in Mediterranean peri-urban forests.* Estudiar la importància dels grups de variables anteriorment esmentats sobre la composició florística de les clapes de bosc de la plana del Vallès. També analitzar la resposta individual de cada espècie per tal d'identificar possibles espècies indicadores de les variables estudiades, així com comparar el paper de les variables ambientals, humanes i estructurals a l'hora de determinar el conjunt d'espècies en aquests boscos.

Capítol 3: *Comparing the role of site condition and landscape structure on understory species richness in peri-urban Mediterranean forests.* Determinar si la riquesa específica de la flora vascular del sotabosc, així com la de diferents sub-grups ecològics i de raresa, ve determinada per variables lligades a l'hàbitat o bé per característiques estructurals del bosc i del paisatge. Es fa especial èmfasi en la importància de l'escala de cada mesura, així com en els efectes pertorbadors antròpics.

Capítol 4: *Landscape and site correlates of human disturbance in peri-urban fragmented forests.* Analitzar, de forma molt preliminar, les preferències antròpiques a l'hora de gestionar i freqüentar les clapes de bosc peri-urbanes en relació a les característiques estructurals d'aquestes.

Capítol 5: *Understory plant species richness and composition in metropolitan forest archipelagos: Effects of forest size, adjacent land use and distance to the edge*. Determinar l'efecte de la mida de la clapa de bosc, dels usos del sòl adjacents (camps de cultiu i àrees urbanes), de la distància al marge del bosc i de la interacció d'aquests tres factors sobre la riquesa i la composició florística del sotabosc. Amb aquest propòsit, es testa l'efecte d'aquests factors sobre la riquesa total d'espècies per parcel·la i sobre els principals grups ecològics i de raresa.

Cal remarcar que en la selecció de les parcel·les s'han exclòs expressament els 10 primers metres des del marge del bosc per tal de limitar l'efecte vora eliminant de l'estudi totes aquelles formacions herbàcies i arbustives del marge del bosc. També destacar que en tots els capítols del present treball la mida de la parcel·la considerada ha estat constant (10 x 10 m^2), ja que s'ha considerat una àrea de mostreig representativa de la flora d'ecosistemes forestals. Els valors de riquesa florística, total i dels diversos grups, que es presenten sempre es refereixen a unitats de mostreig de superfície fixa.

Com es pot observar, l'estudi s'estructura en cinc capítols: els quatre primers es limiten exclusivament als fragments de bosc de la plana del Vallès, mentre que el darrer capítol també té en compte les grans àrees forestals que tenen continuïtat amb les serralades adjacents. Per aquest motiu, els quatre primers capítols es basen en el mateix mostreig, mentre que el darrer és totalment independent. A l'hora d'evitar repeticions innecessàries en els quatre primers capítols, les seves parts comunes s'han redactat de forma més als capítols 1 i 2, mentre que a la resta de capítols tan sols s'han incorporat aquells detalls o variacions que han resultat imprescindibles per a la seva comprensió.

Chapter 1

Quercus and *Pinus* cover are determined by landscape structure and dynamics in peri-urban Mediterranean forests

ABSTRACT

Successional dynamics in Mediterranean forests have been modulated by anthropogenic disturbances during thousands of years, especially in areas densely populated since ancient times. Our objective is to determine whether pine tree cover (early-successional species) and oak tree cover (late-successional species), used as a surrogate of successional stage of peri-urban fragmented forests in the Vallès lowlands (Catalonia, NE Spain), are primarily determined by (1) physiography; (2) anthropogenic disturbances; (3) patch structure; or (4) patch dynamics from 1956 to 1993. *Quercus* spp. and *Pinus* spp. tree cover were separately recorded on 252 randomly-selected plots of 100 m^2, within forest patches ranging in size from 0.25 to 218 ha. Stepwise multiple regressions indicated that forest patch history is the most important variable determining oak and pine tree cover: new forest patches showed higher pine and lower oak tree cover than those patches existing in 1956 that became fragmented from large forest areas (recently split patches). Moreover, patches of 1993 already existing as such in 1956 (unchanged patches) showed higher pine cover than recently split patches. Furthermore, the more connected were forests patches, the higher cover of oak and the lower cover of pines were observed. Finally, highly frequented forests were related to high cover of pines. Physiography variables were not significant. We conclude that pine and oak cover in peri-urban forests are determined by recent patch dynamics, but also by patch spatial pattern. However, man-induced disturbance can modulate this as there is some evidence for pine being associated to a high human frequentation.

INTRODUCTION

The study of correlates determining forest vegetation dynamics becomes especially difficult in Mediterranean forests, due to the intense and complex interaction between anthropogenic factors and Mediterranean landscapes over thousands of years (Barbéro et al. 1990; Zavala et al. 2000). Pure and mixed stands of oak (specially *Quercus ilex*) and pines (specially *Pinus pinea* and *P.halepensis*) dominate extensive forested areas in the western Mediterranean (Quézel & Bárbero 1992; Romane & Terradas 1992; Rouget et al. 2001). In the absence of disturbances, succession generally occurs from pine (shade-intolerant and fast-growing species) to oak (shade-tolerant and slow-growing species) stands in these forests (Terradas 1999). However, in highly-disturbed landscapes, it has been suggested that there is no ‘natural or potential state’ as a valid point of reference (Naveh 1975; Ruiz de la Torre 1990).

Studies carried out at small or medium scales give variable results: while some indicated that habitat disturbances favour pines but they are the major constraint for oak establishment and spread (Gil *et al.* 1990; Morla 1993; Espelta *et al.* 1995; Retana *et al.* 1999; Broncano *et al.* 2005), it is widely known the capacity of oaks to resprout after mechanical disturbances or fire which could lead them to dominate in frequently disturbed stands (Giovanni *et al.* 1992; Zavala *et al.* 2000). At a larger scale, physical and climatic factors have been identified as the most important determinants of the distribution of Mediterranean oak and pine species in Catalonia (NE Spain; Rouget et al. 2001; Thuillier et al. 2003).

Forest tree species composition and dominance can be affected not only by physical disturbances, but also by the pool of propagules able to reach forest patches in fragmented landscapes (Cadenasso & Pikett 2001; Pauchard & Alaback 2004). The lack of nearby propagule sources can reduce the rate of forest recovery in fragmented landscapes, and this leads to a dominance of early successional forests (McClanahan 1986a, 1986b; McClanahan & Wolfe 1993). Several studies have considered simple measures of patch and landscape structure (such as patch size, shape or isolation) as determinants of the pool of species that can colonize forest patches (e.g. Petit el al. 2004). Forest patch history (long-term changes in land use and forest patch attributes) might also affect vegetation dynamics in determining the presence of tree species. However, the relationship between forest patch history and vegetation dynamics has been poorly studied (Di Pasquale et al. 2004), probably because it is often difficult to document and analyse such variables (Debussche et al. 1999).

Our study aims to elucidate the importance of (1) physiography, (2) anthropogenic disturbance, (3) patch structure, and (4) patch historical dynamics on *Quercus* and *Pinus* tree cover, which is considered a surrogate of forest successional stage. The study was conducted in the Vallès lowlands (Barcelona), an example of a peri-urban area with fragmented forests. We hypothesize that the spatial structure (size and distribution) and temporal dynamics (historical changes in patch attributes) of forest patches can be as important in determining the successional status of these forests as disturbances directly induced by humans.

MATERIALS AND METHODS

Study site

The study was conducted in the Vallès lowlands, in the NE sector of the metropolitan area of Barcelona (Catalonia, NE Spain; Fig. 1) and covers an area of 35,478 ha. The proximity to the Mediterranean Sea creates a maritime climate. Mean temperature decreases and rainfall increases to the north.

Major land-use changes have occurred during the last decades in this area (Marull & Mallarach 2005), as a result of a gradual increase in industrial and urban areas, as well as crop abandonment. These changes have created contrasting responses as regards to natural vegetation, with the coexistence of forest fragmentation and recovery processes (Basnou et al. 2003). In 1993, the study site was a complex mosaic of croplands (53% of the area), forests (31%), urban and industrial areas (12%), transport infrastructures (3%) and others (1%). Lowland forests were distributed in patches of up to 490 ha (mean 3.32 ha; ± s.d. 17.40). In contrast, more homogeneous landscapes, made up of large forest areas spanning hundreds or thousands of hectares, dominate the adjacent mountain ranges (Gracia et al. 2000). Consequently, this area is very suitable for analysing the effects of human disturbances and patch dynamics on vegetation structure, since it is a highly man-modified area and exhibits a mosaic of forest patches of variable size.

The study was limited to the Vallès lowland, which was separated from the adjacent ranges and their foothills using a slope threshold of 20º. The elevation of the sampling plots ranged from 115 to 485 m a.s.l.

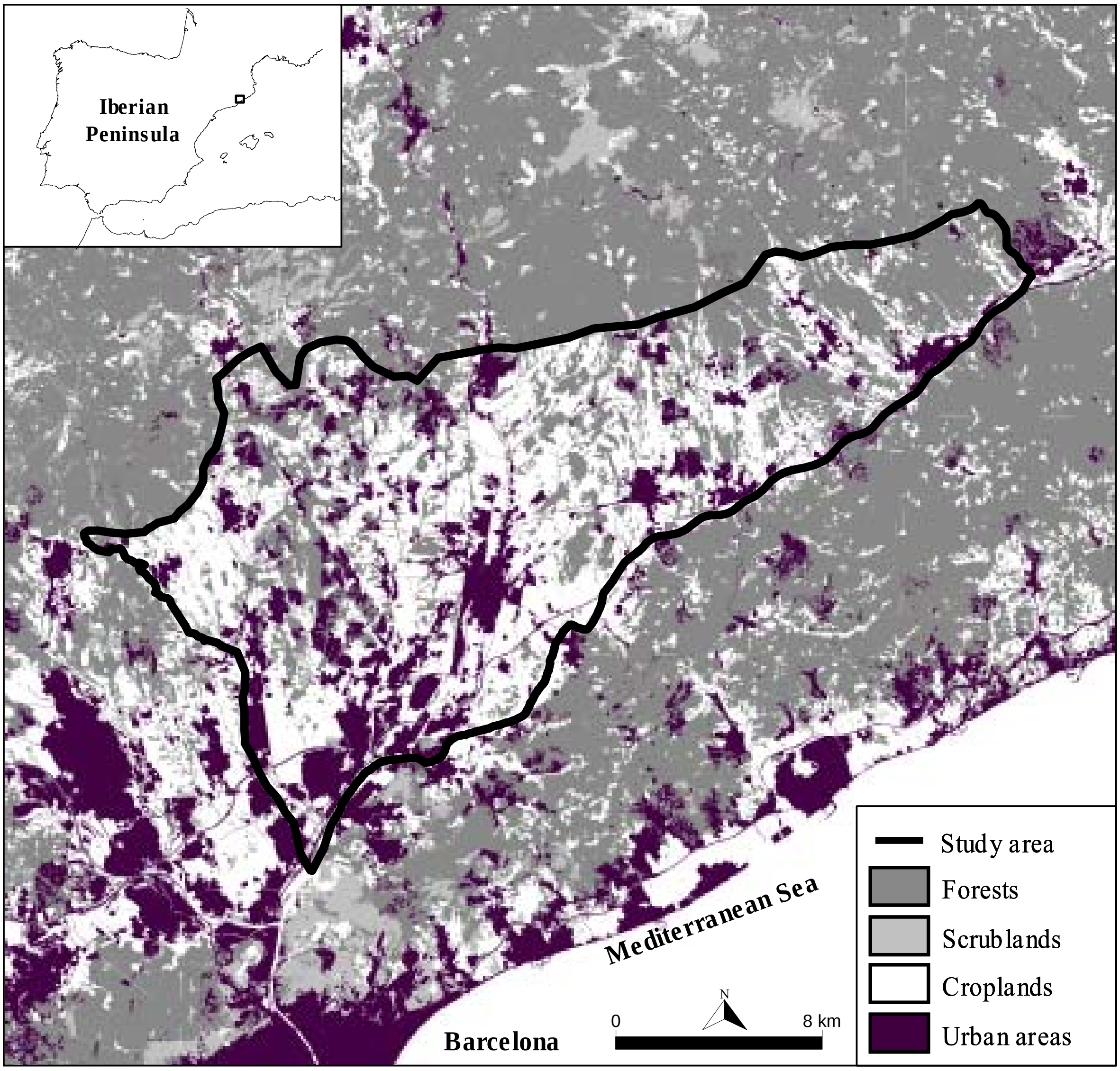

Figure 1. Location of the study area in the northern sector of the Barcelona conurbation.

Plot selection

A geographical information system (GIS) was used to randomly select sample plots from the Land Cover Map of Catalonia (LCMC, CREAF-DMA 1993), a digital map generated by on-screen photo interpreting of 1:25.000 ortho-photomaps. Forest patches isolated from forests on the adjacent mountain range were selected from the LCMC in a separate coverage, totalling 4,379 ha distributed over 1,465 forest patches. In this selection, we did not consider: (1) large forest areas from the adjacent mountain range; (2) forest plots located at the limit of the study site that have been cut during the selection process; and (3) forest patches larger of 250 ha, since there was only one. Finally, most of the forest patches considered (the 93%) had an area <7 ha, whereas the rest of forest patches (the 7%) had an area between 7 and 250 ha. Another coverage with 252 randomly distributed sampling points was then generated on the forest polygons, and in this way a total of 147 forest patches were sampled, ranging in area from 0.25 to 218 ha. Before, we have purposefully excluded the narrow outer belt (0-10 m) of each forest patch where edge effects are trivially stronger. The mean number of plots per patch was related to patch area (r^2=0.567; P<0.0001). All these tasks were performed using MiraMon, a GIS developed at CREAF (Pons 2000).

Data set

In each sampling plot, the cover percentage in the tree layer of each tree species was visually estimated using 10% intervals in an area of 10 x 10 m^2. We considered as a tree those woody plants of more than 2 m, or with a stem diameter at chest height of 10 cm or more. We finally added up, separately, the cover percentage of all *Quercus* (mainly the evergreen *Q. ilex* and the deciduous *Q cerrioides,* but occasionally also the evergreen *Q. suber* and the deciduous *Q. humilis*) and *Pinus* species (mainly *P. halepensis* and *P. pinea,* but also *P. nigra,* and *P. pinaster*).

A set of potential correlates of *Quercus and Pinus* cover per plot was set up from variables related to physiographic conditions, direct and potential human-induced disturbance, and landscape structure and dynamics. Except where indicated, the majority of these variables were calculated from digital coverages using the GIS MiraMon.

Physiography variables included mean annual temperature and rainfall, obtained from the Digital Climatic Atlas of Catalonia (http://magno.uab.es/atles-climatic/index_us.htm; Ninyerola *et al*. 2000), which has a spatial resolution of 180 m. The elevation, slope, and appearance of each plot were obtained from the Digital Elevation Model (DEM, 30 m resolution), generated by the Cartographic Institute of Catalonia (ICC). Geographical position variables (UTM X and Y) were also included in the analysis, thus establishing a trend surface analysis on the regression model (Burrough & McDonell 1998) to account for the spatial variability caused by regional gradients and spatial autocorrelation.

Direct human-induced disturbances were assessed by mean of visual inspections at each plot during field sampling. Two types of disturbances were considered: degree of human frequentation and time since the last forest management. In keeping with Gibb and Hochuli (2002), human frequentation was scored as: low frequentation**,** with little or no evidence of trampling and low to moderate presence of refuse; or high frequentation**,** when both trampling and refuse were moderate to high. The last forest management was scored as: old or non-existent management, when there were no traces of cutting or only cut stumps visible; and recent management, when we found cut branches.

Potential human accessibility was inferred by: (i) the distances between plots and the nearest main road, secondary road and unpaved road, all of them extracted from the 1: 50,000 topographic maps of Catalonia (ICC); and (ii) Forest Accessibility Index (P_i), an aggregate measure of how many people living within a radius of 50 km can potentially access each plot:

$$P_i = \sum_{i=1}^{n} \frac{W_j}{d_{i\,j}^{2}}$$

where W_j is the population of city j; d_{ij} is the distance between plot i and city j; and n is the number of cities in the chosen radius (50 km). It was computed for each plot using the accessibility application of the GIS ArcInfo 7.1.

Finally, we established the following variables for landscape structure: (i) forest patch size; (ii) the percentage of major land uses surrounding each forest patch (roads, urban areas, shrublands-grasslands, and crops); and two variables designed to estimate the connectivity of each forest patch: (iii) forest area within a radius of 500 m around the patch perimeter, and (iv) the total area of those forests that stand, either wholly or partially, within this 500-m radius. The landscape dynamics were inferred from forest patch history over the period from 1956 to 1993, which was studied by combining the LCMC forest boundaries (1993) with those of a land cover map from 1956, generated *ad hoc* from aerial photographs that were georeferenced, assembled and photo-interpreted. Forest patches from 1993 were then scored as: (a) already existing as such in 1956 (‘unchanged patches’); patches fragmented from large forest areas existing in 1956 (‘recently split patches’); and patches existing in 1993 but not in 1956 (‘new patches’).

Statistical analyses

The association of all the explicative variables with oak (*Quercus* spp.) and pines (*Pinus* spp.) tree cover was analyzed by separate stepwise multiple regression analyses. Pearson’s correlation coefficients were previously performed for each pair of continuous variables in order to reduce collinearity. The threshold value for deciding on redundancy was set to a correlation coefficient of $|r|=0.75$. As a result, mean annual temperature, mean annual precipitation, and the percentage of crops surrounding forest patches were rejected because they were highly correlated with altitude, UTM X and the percentage of urban areas, respectively ($|r| >0.75$, $P< 0.05$).

We performed a backward stepwise regression analysis on each dependent variable, using the standardised beta coefficient as the ordination criterion for the importance of each predictive variable. The distribution of all the dependent variables was normal, after performing, where necessary, an arcsin transformation. Categorical variables were coded as zero/one dummy variables and were included in the regression model. In the case of forest history, made up of three levels, we did not include the ‘new forest patch’ level after the dummy codification, to avoid collinearity.

Dependent variables could be influenced by spatial autocorrelation. This correlation is a problem in standard statistical tests when data points are not independent, as this contradicts the basic assumptions of statistical analysis (Fortin et al. 1989; Legendre et al. 2002). To determine the degree of spatial autocorrelation in our data, we examined Moran’s I coefficients on the standardised residual of each model (Cliff & Ord 1981). The results showed high values of autocorrelation for both models (–0.66 for *Pinus* spp. and 0.98 for *Quercus* spp., respectively). Following other studies (e.g., Deutschewitz et al. 2003; Pino et al. 2005), we used the modified *t*-test of Dutilleul (Legendre 2000) to reduce the level of autocorrelation. This test takes spatial autocorrelation into account when calculating the degrees of freedom, and corrects the tests for the number of significant regressors (Legendre et al. 2002). However, this method is only feasible for continuous variables. To correct the autocorrelation in the case of dummy variables, we carried out simple regressions for each independent *versus* dependent variable, using 10 random sub-samples of 50% of the total sampling points. We determined as significant those dummy variables that reached *P*-values <0.05 for the 10 simple regressions.

We also tested the association of forest patch history with forest connectivity measures by means of one-way ANOVA. Pairwise comparisons were performed for the significant factors and interactions, using the Tukey test.

RESULTS

Altogether, we recorded 10 different tree species for the 252 sampling plots: four of these were oaks, four were pines and the other two species were *Sorbus domestica* and *Ulmus* *minor*, found only occasionally. The most represented species were *Quercus ilex*, followed by *Pinus pinea* and *P. halepensis*, respectively (Table 1).

Among the categorical variables tested, low human frequentation was more frequent than high human frequentation (144 *versus* 108 plots respectively). Moreover, recent forest management predominated over old management (146 *versus* 106 plots respectively). Finally, most of the studied plots (207) were located in ‘unchanged patches’, whereas much fewer corresponded to ‘new patches’ or to ‘recently split patches’ (28 and 17 plots, respectively).

The regression model obtained for the cover percentage of *Quercus* spp. captured a low proportion of the total variance (r^2= 0.21; P<0.0001; Table 2). Seven of the 19 explicative variables tested were significantly correlated with *Quercus* cover. However, after correcting for spatial autocorrelation, only 3 variables remained significant. Of these, ‘unchanged patch’ was the first correlated variable, followed by ‘recently split patch’. In both cases, the percentage of oak tree cover was higher than for ‘new patch’ (Fig. 2). Finally, oak tree cover increased with the ‘total area of forest standing, either wholly or partially, within 500 m around the patch perimeter’, meaning that more isolated forests had less oak tree cover.

Table 1. Mean cover (%) per plot (± s.d.) of the tree species recorded in the study.

Tree species	**Cover (%)**
Quercus ilex	46.00 ± 31.75
Pinus pinea	22.27 ± 30.48
Pinus halepensis	19.28 ± 27.14
Quercus cerrioides	12.17 ± 19.37
Pinus nigra	1.57 ± 10.31
Pinus pinaster	1.01 ± 7.46
Quercus suber	0.24 ± 2.38
Quercus humilis	0.16 ± 2.06
Sorbus domestica	0.08 ± 1.27
Ulmus minor	0.04 ± 0.63

Also, a low proportion of total variance was captured by the regression model obtained for the percentage of pines tree cover ($r^2 = 0.19$; $P<0.0001$; Table 2). Six of the 19 explicative variables were significantly correlated with pine cover. After correcting for spatial autocorrelation, 3 of them again proved significant: ‘recently split patch’ was the first correlated variable, followed by ‘forest frequentation’ and ‘forest area within 500 m around the patch perimeter’. ‘Recently split patches’ showed lower percentages of pine cover than other patch history types (Fig. 3). Moreover, highly frequented forests were related to high cover of pines. Finally, pine cover increased when ‘the forest area within 500 m around the patch perimeter’ decreased, meaning that pine tree cover increases with forest patch isolation.

Finally, ANOVA results showed that ‘the total area of forests standing either wholly or partially within 500 m around the patch perimeter’ and ‘the forest area within 500 m around the patch perimeter’ were not significantly related to forest patch history (F=1.87 and F=2.40 respectively; P>0.05).

Table 2. Results of stepwise linear regression analysis for (a) *Quercus* spp. and (b) *Pinus* spp. tree cover (BETA: standardized regression coefficient; *t*: t-statistic; *P*: significance) and results after testing for spatial autocorrelation using Dutilleul's modified *t*-test for continuous variables (t_{mod}: modified *t* statistic, df_{mod}: modified degrees of freedom; p_{mod}: significance) and after regression analyses for dummy variables (t_{range}: lower and higher value of the t-statistic of the repeated analyses; *df*= lower and higher value of the degrees of freedom; *p*: significance).

Dependent variables	**BETA**	***t***	***P***	$\mathbf{t_{mod}}$	$\mathbf{df_{mod}}$	$\mathbf{P_{mod}}$	$\mathbf{t_{range}}$	$\mathbf{df_{range}}$	***P***
(a) *Quercus* cover (r^2= 0.21; *P*<0.0001)									
UTM X	-0.316	-3.026	**	0.053	257.134	n.s.			
UTM Y	0.426	3.200	**	0.854	193.374	n.s.			
Total area of forests wholly or partially within 500 m[1]	0.171	1.872	*	3.780	213.208	*			
Altitude	-0.399	-3.647	**	0.276	214.360	n.s.			
Human frequentation	-0.140	-2.167	*	-	-	-	-1.754; -1.670	117; 126	n.s
Patch history-I[2]	0.282	3.633	***	-	-	-	2.023; 2.847	120; 124	**
Patch history-II [3]	0.290	3.688	***	-	-	-	2.850;3.123	120; 124	**
(b) *Pinus* cover (r^2= 0.19; *p*<0.0001)				-	-	-			
UTM Y	-0.337	-2.501	*	2.699	64.650	n.s.			
Forest area within 500 m	-0.162	-1.682	*	6.570	214.140	*			
Human frequentation	0.172	2.616	*	-	-	-	1.980; 2.479	122; 137	*
Distance to the main roads	0.177	2.158	*	1.346	102.386	n.s.			
Patch history-I[2]	-0.205	-2.604	**	-	-	-	-1.995; -2.073	122; 124	*
Patch history-II [3]	-0.166	-2.074	*	-	-	-	-1.584; -0.506	120; 122	n.s

Significance of variables: n.s, non significant; *, <0.05*;* **, <0.01; ***, <0.001.

[1] Total area of forests located, wholly or partially, within a radius of 500 m around the patch perimeter.

[2] Forest patches split from 1956 large-forest areas.

[3] Forest patches existing as such in 1956.

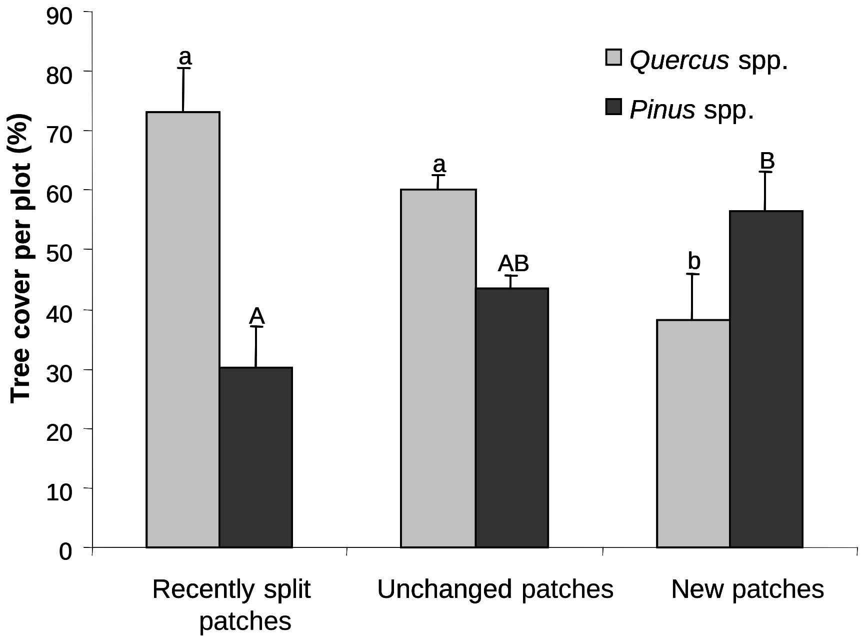

Figure 2. Mean *Quercus* and *Pinus* tree cover per 100 m^2 plot as a function of recent patch history. Error bars are one standard error of the mean. Different capital letters indicate significant differences for *Pinus* spp.; different lower-case letters indicate significant differences for *Quercus* spp.

DISCUSSION

Forest patch history has been the most important variable in the determination of oak and pine tree cover in the Vallès lowlands (Fig. 2). In general, our results agree with the expected patterns of succession in Mediterranean forests: pines easily colonize recently abandoned areas (Acherar et al. 1984; Lepart & Debussche 1992; Debussche et al. 1999), whereas oaks substitute pines in later stages, when the habitat conditions suit them better (Bárbero et al. 1998; Terradas 1999). What is new in our approach is the link we can establish between patch history and successional stage in terms of pine/oak cover. Some of the previously cited studies have also highlighted the importance of spatial patterns and historical processes in vegetation dynamics in Mediterranean landscapes (e.g., Lepart & Debussche 1992; Debussche et al. 1999).

The 'unchanged patches' category occupies an intermediate successional stage between 'new patches' and 'recently split patches', probably because it makes up a mixture of historical situations, including new forests originated *ex novo* or split from large areas just before 1956, and unchanged patches already existing as such for a considerable time. These 'unchanged patches' probably display complex historical dynamics, including oak progression and regression phases, since they are more prone to processes associated with fragmentation than large areas, to a great influence from adjacent land use and to direct human-induced disturbance, as we demonstrate in chapter 5. Human disturbances create open areas where light availability increases (Hobbs 1988; Gilliam 1995; Deconchat & Balent 2001), favouring the arrival and persistence of pine seeds (shade-intolerant species; Bárbero et al. 1998). In contrast, forest tree conditions might still persist in 'recently split patches', since they were separated only a few years ago, while forest succession dynamics might require longer periods of time. We only considered historical patch dynamics related to the last 40 years, but previous processes might also have influenced the current situation. Longer-term approaches are needed in future research to elucidate the origin and dynamic of such forest patches, especially for 'unchanged patches'.

Habitat connectivity has been identified as one of the most important factors with respect to the possibility of population dispersal (Taylor et al. 1993; With et al. 1997). We found that highly-connected forests facilitate advanced successional stages, whereas more isolated forests have more pine tree cover. We attributed this to the fact that oaks are short-distance seed dispersal species in comparison to pines, as suggested by Zavala & Zea (2004). We suggest that the successional stage of forest patches in the Vallès lowlands not only depends on patch historical dynamics, but also on the current landscape structure. However, other studies have obtained contradictory results after studying these species' seed dispersal distances; for example, Acherar et al. (1984) found that dispersal distances in pines were short in the forests of Languedoc (France); they determined that 97% of *Pinus halepensis* seeds were dispersed less than 24 m from the tree of origin. Moreover, Gómez (2003) found that the distance of holm-oak acorn dispersal by jays (*Garrulus glandarius*) was high in the Sierra Nevada Protected Area (SE Spain); he found that acorn dispersion had two peaks: one at a short distance (1-20 m), within the same oak patch, and another produced by long-distance dispersal to other forests, generally pine stands (200-500 m). Further research is needed to ascertain the relationship between forest fragmentation and seed dispersal in highly-humanised areas, especially with respect to the behaviour of animal seed dispersers (*Sciurus vulgaris, Garrulus glandarius*) across a highly man-modified matrix. Other historical factors related to human activities, but not considered in the present study, could also have modulated the relationship between landscape structure and species distribution over decades on a landscape scale in the Vallès (e.g., setting aside the less accessible forests on mountain slopes for charcoal production, thus favouring oak resprouting, or cultivating pines for nut extraction in isolated patches on lowlands); these could have important consequences in the current situation.

Our study also provides the interesting result that patch history is independent of connectivity in the Vallès. Despite the fact that ‘recently split patches’ are more advanced successionally and ‘new patches’ are generally in an earlier successional stage, the finding that forest connectivity was not related to forest patch history suggests that ‘recently split patches’ may suffer successional regression in the future if they are highly isolated, whereas new patches may further their successional stage quickly if they are highly connected.

Human activities have had important effects on Mediterranean forest ecosystems for thousands of years (Grove 1996; Verdú et al. 2000; Gondard et al. 2001). Anthropogenic pressures on forest plots have been identified as important correlates of understory species richness in the Vallès lowland forests (see chapter 3). However, the anthropogenic variables studied in the present study (both direct human-induced disturbances and potential human accessibility) did not have a determining effect on oak and pine tree cover, probably because trees mainly respond to longer time-scale processes (e.g., Thuiller et al. 2003). Previous studies of oak forests have demonstrated that 25-30 years are required to reach canopy closure after disturbances (Espelta et al. 1995), whereas our direct-human induced disturbances cover shorter periods of time. We have found that only pine cover increased with high human frequentation. It is known that pines recruit seedlings in the first years after disturbance because there is greater availability of light, whereas oaks require long periods free of disturbance to successfully establish seedlings (Zavala et al. 2000). However, the relationship between pines and human frequentation could run in both directions: as we suggest in a previous study (Guirado 2002), human frequentation could be favoured by pine trees rather than by oak forests, since the former are more accessible for transit purposes. Our study also highlights the absence of any relationship within the study area between pine and oak cover, on the one hand, and physiography, on the other. Altitude was found to affect Mediterranean forest-tree dynamics in several large-scale studies (e.g. Rouget et al. 2001; Thuiller et al. 2003), but not in the Vallès lowlands, where the elevation gradient is relatively small. Similarly, and despite the fact that slope and aspect have been considered important factors in the determination of vegetation dynamics in previous studies (e.g., Didham & Lawton, 1999; Gracia et al. 2002), such variables were not considered significant in our case, as we studied only the lowland areas of the Vallès, where the topography is generally gentle. We can conclude that the physical variables studied in the Vallès lowlands are relatively homogeneous in relation to forest tree cover responses.

In short, a long period of human intervention in peri-urban Mediterranean landscapes has resulted in considerable uncertainty as regards the causes of forest succession. In such areas, it is important to consider the role of anthropogenic disturbances and spatial processes to describe succession. In the present study, we highlighted the importance of forest dynamics in a medium-length time period (over 40 years) and landscape structure (patch connectivity) to determine the successional stage of these forests. However, uncontrolled historical activities produced in this area over decades or centuries might have an important role in determining the current situation.

Chapter 2

Sensitivity of understory species to physiography, man-induced disturbance and landscape structure in Mediterranean peri-urban forests

ABSTRACT

Indicator species have increasingly been used as a useful tool for determining forest condition and human disturbances. The objective of the present study was to investigate the response of physiography, man-induced disturbances and landscape structure and dynamics of understory plant species in peri-urban fragmented forests. The presence of all vascular plants was recorded on 252 plots of 100 m^2 randomly selected within 147 forest patches ranging in size from 0.25 to 218 ha. We performed a Canonical Correspondence Analysis (CCA) to explore the multivariate association between plant species composition and the complete array of variables. We also carried out specific analyses (Indicator species analyses and Generalized Linear Models) to detect indicator species for each variable studied. CCA showed an ordination along the main axis related to forest disturbance, and the existence of three main groups of understorey species associated to: well preserved forests, highly connected forests and more disturbed forests. Sixty-eight percent of the species responded to, at least, one of the variables analyzed. More species responded to site conditions than to landscape variables. *Quercus* spp. tree cover was the variable affecting most species, especially those most dependent on a closed-canopy environment. Moreover, a lot of the species were sensitive to anthropogenic disturbances. General interpretation of these results enables the identification of ecological groups which are particularly sensitive to forest disturbances, and should be considered in developing further conservation policies.

INTRODUCTION

Plant species composition in a given landscape is determined by a complex web of factors: physical attributes (climate, soil, topography, altitude, moisture), natural disturbances, life history traits, competition, and many others. Besides the natural environmental factors, anthropogenic disturbances can be a major constraint determining plant species distribution in forests. This is especially evident in the Mediterranean basin, where direct and indirect human activities have been the most important landscape modulators for millenia (Barbéro et al 1990; Lepart & Debussche 1992; Roche et al. 1998; Rouget et al. 2001; Gondard et al. 2003). Modifications of forest composition, structure and dynamics are especially evident in urban and peri-urban landscapes, mainly due to urban development, the intensively used landscape matrix and the high recreation pressure (Honnay et al. 1999b; Godefroid & Koedam 2004a,b). In consequence, peri-urban Mediterranean forests in particularly have suffered great changes in their extent, structure and species composition directly through human disturbance, or indirectly through land-use changes and other anthropogenic influences (di Castri 1981; Suc 1984; Mooney 1988). These changes might lead to a decrease in native species and an increase of exotic and ruderal species (e.g., Bárbero et al. 1990; Gondard et al. 2003). For example, land use changes and forest management may negatively affect forest plants through their impact on soil characteristics (Dumortier et al. 2002; Godefroid & Koedam 2004a); in addition, forest patches may be invaded by non-forest plant species coming from the surrounding man-modified habitats (Murcia 1995; Honnay et al. 1999b; Hobbs 2000; Godefroid & Koedam 2003b), also roads may act as a vector of dispersion of alien plants into the forest patches (Forman et al. 2000, 2003; Godefroid & Koedam 2003a, 2004b; Pauchard & Alaback 2004). Forest patch isolation due to urban development may also enhance ruderal and exotic species establishment through its impact on propagule pressure (Dzwonko & Loster 1992; Grashof-Bokdam 1997; Honnay et al. 1999a).

Because of the complexity of these processes, environmental managers need a basic understanding of the potential effects of human activities on forest ecosystems (Dale et al. 2002). Indicator species have increasingly been used as a tool to monitor forest condition (Mendoza & Prabhu 2003; Moffatt & McLachlan 2004). There are many examples in which understorey vegetation has been used as an effective indicator of ecological conditions (e.g., Ter Braak & Gremmen 1987; Hoffmann 1998; Hermy & Cornelis 2000; Wulf 2001; Bio et al. 2002). Plant species might also be effective indicators of forest disturbance, since many of them are highly susceptible to changes in site conditions and are relatively easy to detect and to identify (Hoffmann 1998). However, despite the negative effect of human activities on peri-urban forests, few studies identify indicator species of direct anthropogenic disturbances in site conditions (e.g., Sutter et al. 1993; Dale et al. 2002; Moffatt & McLachlan 2004), and much less distinguish indicator species of forest patch structure and landscape dynamics as a result of human activities (e.g., Moffatt & McLachlan 2004).

In chapter 3, we will find that there is a complex set of factors measured at contrasting scales that affect understorey species richness in fragmented forests of the Vallès lowlands. We demonstrated that most of the variance explained was related to site conditions and landscape structure variables measured at plot level. *Quercus* tree cover was the most important factor explaining non-forest species richness, which decreased when *Quercus* tree cover increased. Moreover, human-induced disturbance was an important correlate of synanthropic species (i.e. those thriving in man-made or man-disturbed habitats) and total species richness, which were both higher in recently managed and highly frequented forests. The present paper aims to explore these associations at species level. To do that, we investigate the individual response of understorey plant species to physiography variables, man-induced disturbance, and landscape structure and dynamics with the objective of identifying indicator species. We first performed a Canonical Correspondence Analysis to explore the multivariate association between plant species composition and the complete array of independent variables for 252 sampling plots randomly placed in the study area. Secondly, we performed specific analyses to detect indicator species for each variable studied. As the analysis was performed simultaneously for all species and variables, we finally assessed the comparative role of environment, man-induced disturbance and landscape structure in determining plant species assemblage in these forests.

MATERIALS AND METHODS

The area and plot selection procedures have been described in chapter 1.

For each sampling plot (n=252), we recorded all understorey vascular plants in an area of 100 m^2 as dependent variables. Species' presence was then tested against a set of independent variables classified in two main groups (Table 1): (i) site condition variables, measured exclusively at plot level and including physiography and direct-human induced disturbances; and (ii) landscape structure and dynamics which were measured at plot or at patch level.

Physiography variables included physical attributes (latitude, longitude, altitude and slope), and the percentage of *Quercus* ssp. and *Pinus* spp. tree cover. Direct human-induced disturbances were represented by human frequentation and time since the last forest management (LFM). Landscape metrics at plot level included the following variables related to potential human accessibility: (1) distance from the centre of each plot to the forest edge; (2) distances to the nearest main road, secondary road and unpaved road; and (3) the Forest Accessibility Index.

We also considered the following landscape metrics calculated for each forest patch hosting sampling plots: (1) patch size (in ha); (2) patch shape, assessed using the mean patch fractal dimension index (MPFD, Elkie et al. (1999); (3) surrounding land uses (the percentage of patch perimeter shared with each of its adjacent major land uses, *sensu* Hersperger & Forman (2003)). We considered the following land uses: roads and urban areas, shrublands-grasslands, and crops; and (4) the connectivity of each patch, calculated as the forest area within a buffer of 500 m around the patch perimeter. This distance was selected as it is considered a general upper limit of woodland plant dispersal (Van Ruremonde & Kalkhoven 1991; Grashof-Bokdam 1997). Finally, patch history over the period 1956 to 1993 was included as: a) 'unchanged patches'; (b) 'recently split patches'; and (c) 'new patches'. For complete information about the independent variables see chapter 1.

Table 1. Measured independent variables. Abbreviations and minimum and maximum values for each variable are listed.

Variables	Abbreviations	minimum value	maximum value
Site conditions			
A. Physiography			
UTM X coordinates	UTM X	429620	459684
UTM Y coordinates	UTM Y	4601140	4619420
Slope (º)	Slope	1	20
Altitude (m)[*]	Altitude	85	485
B. Successional stage			
Percentage of *Quercus* tree cover[a] (%)	QuercusCover	0	165
Percentage of *Pinus* tree cover[b] (%)	PinusCover	0	110
C. Direct human-induced disturbances			
Forest frequentation (low=1, high=2)	Frequentation	Categorical	
Time since the last forest management (Long ago=1, Recent=2)	LFM	Categorical	
Patch and landscape metrics			
A. Potential human accessibility			
Distance to the edge (m)	EdgeDist	15	283
Distance to nearest main roads (m)	R1	23	8555
Distance to nearest secondary roads (m)	R2	15	1500
Distance to nearest unpaved roads (m)	R3	15	412
Accessibility index (no units)	Acc50	0.005	0.13
B. Spatial metrics			
Forest patch area (ha)	Patch area	0.25	218.5
Mean patch fractal dimension index (non dimensional).	MPFD	1.25	1.5
Connectivity (ha)	connect.	1.72	203.4
Percentage of roads and urban areas surrounding forests (%)	Road-Urban	0	100
Percentage of shrubland-grassland surrounding forests (%)	Shrub-Grass	0	100
Percentage of crops surrounding forests (%)[*]	Crops	0	100
C. Patch history (recently split patches=1, unchanged patches =2, and New patches=3; see text)	Patch history	Categorical	

[a] Cumulative percentage of *Quercus ilex* and *Q. cerrioides* cover.

[b] Cumulative percentage of *Pinus pinea*, *P. halepensis*, *P. pinaster* and *P. nigra* cover.

[*] Variables not considered because they were highly correlated (|r|>0.70) with other variables.

Statistical analysis

All the analyses are based on presence-absence data of species occurring in 4 or more sampling plots. Effects of site conditions and patch and landscape metrics on plant species composition were analysed through Canonical Correspondence Analysis (CCA; Ter Braak 1987) using Canoco 4.5 for Windows (Ter Braak & Šmilauer 2002). This method is appropriate for predicting presence-absence of species at given spatial locations (Guisan & Zimmermann 2000). Pearson correlations among the independent variables were previously checked in order to find highly correlated pairs of variables, and also to avoid collinearity problems. The threshold value for determining redundancy was set to a correlation coefficient $|r|>0.70$. In consequence, altitude and the percentage of crops surrounding each forest patch were rejected because they were highly correlated with UTM Y ($|r|= 0.74$) and with the percentage of roads and urban areas surrounding forest patches ($|r|= 0.84$) respectively. Three explanatory variables were categorical, namely human disturbance of forests (forest frequentation and time since the last management) and patch history, and the other 15 were quantitative. Categorical variables were included as dummy variables into the CCA, rejecting one category for each variable to avoid collinearity.

Indicator Species Analysis (ISA; Dufrêne & Legendre 1997) was performed using PC-ORD v.4 (McCune & Mefford 1999), in order to determine indicator species for the categorical variables studied. Indicator species are defined as the most characteristic species of each factor level, found mostly in a single level and present in the majority of the plots belonging to this level. An indicator value ($IndVal_{ij}$) for species i in level j is computed as the product of A_{ij} (the absolute frequency of occurrence of species i in the level j) by B_{ij} (the relative frequency of occurrence of species i in the level j) multiplied by 100. Finally, the indicator values for each species in each group were tested for statistical significance using a Monte Carlo technique (Dufrêne & Legendre 1997). The indicator values range from zero (no indication) to 100 (perfect indication). It is only possible to run ISA when the number of plots in each level of the studied factor is as similar as possible. Therefore, we only calculated ISA for direct human-induced disturbances, whereas for forest patch history it was not possible to apply this analysis.

Generalised Additive Modelling regressions with a cubic smooth spline function (GAM; Hastie & Tibshirani 1990) were performed to determine the response of each plant to the continuous variables. A conservative significant level (*P* ≤ 0.001) was used to select indicator species because of the large number of tested species and to ensure the usefulness of the species selected. GAM regressions have been used in numerous studies of species-environment relationships (e.g. Bio et al. 1998; Austin 1999; Guisan & Zimmermann 2000; Bio et al. 2002; Godefroid & Koedam 2004a), because they can dela with non-Gausian error distribution and nonlinear relationships between response and predictor variables (Austin & Meyers 1996). These models include a variety of smooth functions that estimate the response for each dependent variable on the responses observed for neighbouring values on the predictor gradient. To simplify the additive models in this study, we restricted each species-predictor response to a curve using a maximum of 3 degrees of freedom. Higher polynomials tend to reveal spurious and biologically unfeasible response shapes that are more difficult to interpret (Austin et al. 1990; Bio et al. 1998). The optimum degree of freedom for each species was selected by means of a step wise selection using the Akaike Information Criterion (AIC; Akaike 1973; Sakamoto *et al.* 1986). The response data were the presence-absence of plant species, and the resulting binomial distribution was adapted with a logistic link function. Generalized additive models cannot be easily summarized numerically, and are best summarized by plotting the estimated smooth terms representing the relation between the values of the predictor and its effect on the modelled response variable (Leps & Šmilauer 2003).

RESULTS

A total of 241 plant species were recorded in the 252 sampling plots. Of these species, those found in four or more plots were included in the study. We also rejected those plants only identified at the genus level. Finally, 96 species were considered in the present study (See Appendix I).

Predictors of plant species composition

CCA shows an ordination along the main axis related to forest disturbance, and the existence of three main groups of understorey plant species characterised by their associated variables (Fig. 1): Species scored on the lower right side of Fig. 1 are characteristic of relatively well-preserved forest stands belonging to late-successional stages, with large *Quercus* ssp. cover (i.e. *Rubia peregrina, Hedera helix,* and *Asparagus acutifolius*). These stands occur normally in marginal steep slope areas, forming patches of high mean fractal dimension index. Species on the upper right side of the diagram mainly correspond to shrubs (i.e. *Juniperus oxycedrus, Phillyrea angustifolia* and *Erica arborea*), are also associated to relatively undisturbed stands, and characterised by spatial metrics such as distance to the forest edge and patch connectivity, and also geographic position. Finally, species on the left side of the diagram are associated to direct or potential disturbance related to high accessibility index, recent forest management and early successional status as estimated from *Pinus* spp. cover.

Species-environment correlations in CCA were 0.725 for the first axis and 0.678 for the second one. The first two axes of the CCA explained 67% of total variance (Table 2) with 16 significant explanatory variables out of the 18 initially tested, according to the Monte Carlo Permutation Test ($P \leq 0.05$). The percentage of *Quercus* tree cover was the most powerful predictor explaining 8% of the floristic variation, followed by the UTM X (7%). The distance to the main roads and patch history (at ‘unchanged patch’ level) explained 5% of the variation. UTM Y, forest management and slope accounted each for 4% of the variation in the understorey, whereas the rest of the landscape metrics each explained 3% of the variation in species assemblage. The degree of forest frequentation and the distance to unpaved roads did not significantly contribute to the overall variation. In summary, 32% of the variation in species assemblage is explained by site conditions and 39% by landscape structure and dynamics.

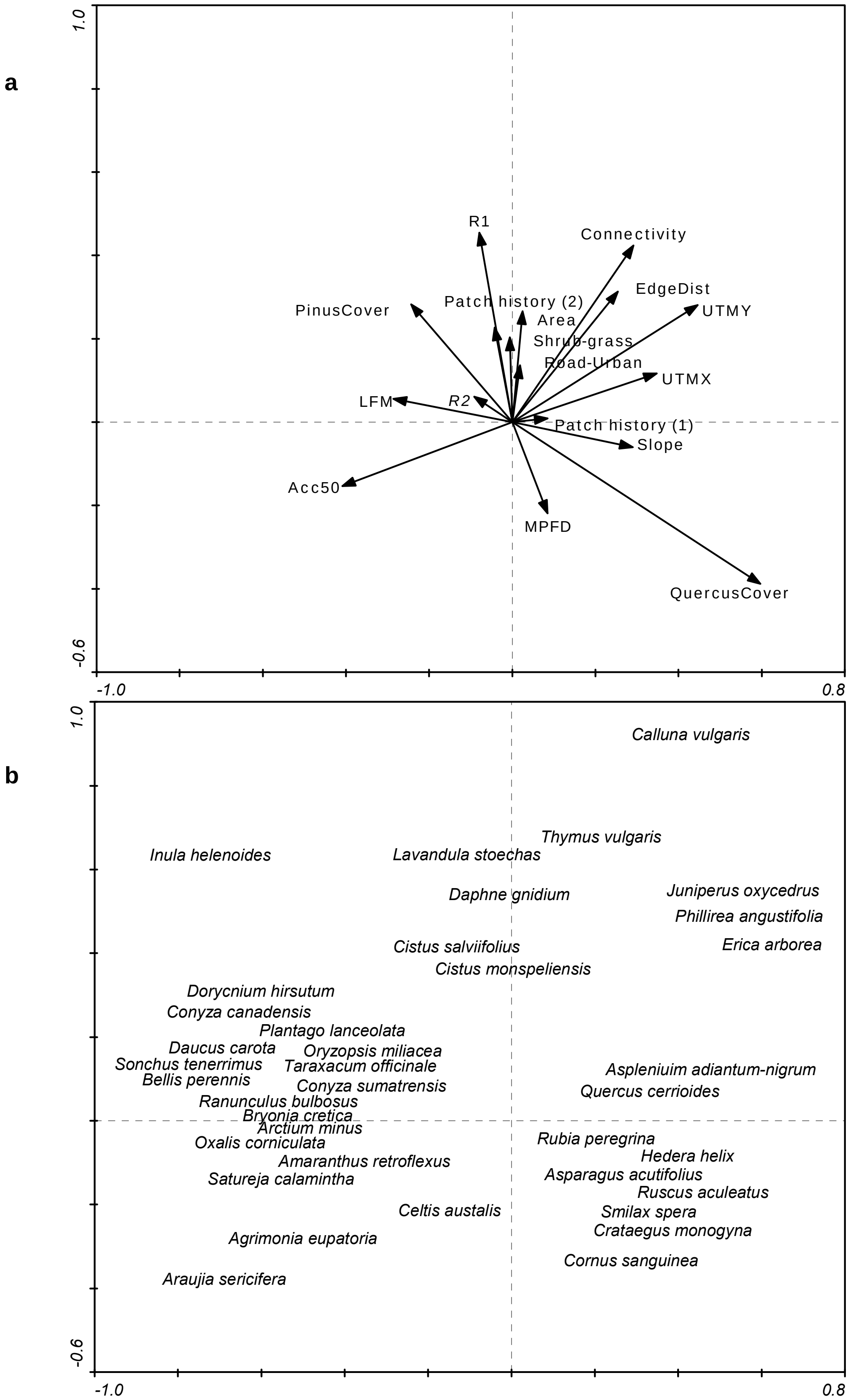

Figure 1. Species ordination diagram based on the Canonical Correspondence Analysis with respect to fourteen quantitative variables and three categorical variables. The axes are scaled in standard deviation units. Eigenvalues of first and second axis were 0.141 and 0.085 respectively. (a) Only significant variables, as well as (b) the most confident species (with explained variance $\geq$ 5%), are represented. Species abbreviations are based on the first four letters of genus and species names. For explanation of variable abbreviations, see table 1.

Table 2. Percentage of variance explained by explanatory variables and level of significance ($P \leq 0.05$; Monte Carlo test).

Explanatory variable	Variance explained by single variable (%)	Cumulative variance explained (%)	*P*-level
Quercus Cover	8	8	0.002
UTMX	7	15	0.002
Distance to the main roads	5	20	0.002
Patch history (unchanged patches)	5	25	0.002
UTMY	4	29	0.002
Recent forest management	4	33	0.002
Distance to the forest edge	3	36	0.002
Slope	4	40	0.002
Pinus Cover	3	43	0.002
Accessibility	3	46	0.032
Patch area	3	49	0.006
Connectivity	3	52	0.002
Percentage Shrub-Grass	3	55	0.012
Patch history (Recently split patches)	3	58	0.010
Mean Patch Fractal Dimension	3	61	0.014
Distance to the secondary roads	3	64	0.016
Percentage Road-Urban	3	67	0.012
High frequentation	2	69	0.096
Distance to unpaved roads	2	71	0.244

Indicator species

Of the 96 study species 68 (70.8%) were indicators of, at least, one of the variables analysed. Fifty five percent of the species indicated site conditions, whereas less species (33%) were related to landscape metrics.

Site conditions

Twenty-two species (23% of the total considered) showed highly significant association with direct human-induced disturbances (Table 3): nine species significantly indicated either long ago or recent forest management, and 16 species were related to either high or low human frequentation. Indicator species reached low indicator values, ranging between 4 and 38 (we only considered the highest indicator value for each factor). Nine species (9.4% of the species considered) had values greater than 15. Among them, *Celtis australis* was the best indicator of long ago forest management, while *Bryonia cretica, Conyza sumatrensis, Satureja calamintha, Taraxacum officinale* and *Viola sylvestris* were the best indicators of recent forest management. Species indicating low forest frequentation were *Oryzopsis milliacea*, and seedlings of *Quercus cerrioides* and *Q. ilex,* while *Taraxacum officinale* indicated highly frequented forests.

Table 3. Indicator values for significant (*P*≤ 0.05) indicator species as identified by the method of Dufrêne and Legendre (1997) for the two categorical variables analysed.

	Time since last forest management		Forest Frequentation	
	Long ago	Recent	Low	High
Araujia sericifera			0	7
Arctium minus	0	11	2	10
Bryonia cretica	2	16		
Celtis austlis	27	15		
Chenopodium album			1	12
Cirsium vulgare			1	8
Conyza canadensis	0	11		
Conyza sumatrensis	3	18		
Coriaria myrtifolia	4	0		
Daucus carota			1	10
Dorycnium hirsutum			1	11
Lithospermum purpurocaeruleum			0	6
Olea europaea			2	10
Oryzopsis miliacea			11	23
Oxalis corniculata			2	14
Plantado lanceolada			1	9
Quercus cerrioides			38	18
Quercus ilex			31	16
Ranunculus bulbosus			2	11
Satureja calamintha	2	17	4	15
Taraxacum officinale	4	16	5	17
Viola sylvestris	4	17		

For continuous variables (Table 4), 26 species (27% of the species considered) were highly significantly associated to physiography (Fig. 2). Nineteen species were related to UTM Y, 13 species to UTM X, and 4 species to slope. None of the species indicated altitude. A*splenium-adiantum nigrum* and *Lavandula stoechas* showed a significant increasing trend when slope increased, while riparian plants such as *Inula helenoides* and *Lithospermum* *purpurocaeruleum* had a narrow response to slope, with optimum values varying from 5 to 10º. For UTM Y (latitude), 5 species decreased and 7 species increased when latitude increased, and 7 species indicated low amplitude of values (e.g., *Sorbus domestica*). Some species showed an increasing (e.g., *Erica arborea*) or decreasing trend (e.g., *Inula helenoides*) with UTM X, whereas others exhibited an optimum for intermediate (e.g., *Araujia sericifera*) or extreme (e.g., *Calluna vulgaris*) values of longitude.

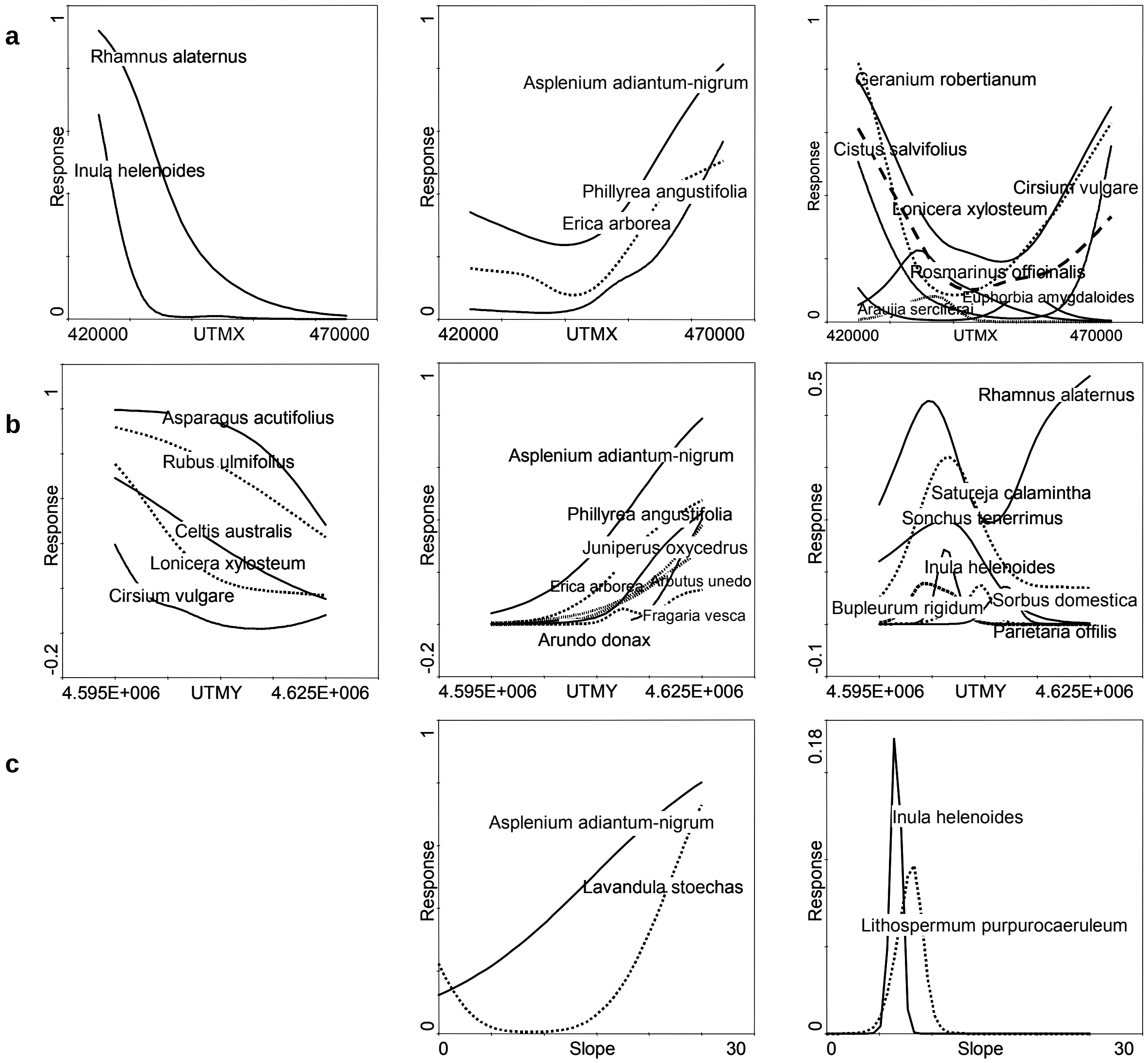

Figure 2. Significant species response ($P \leq 0.001$) for physical attributes: a) UTM X; b) UTM Y; and c) slope. Response axes represent the probability of each species to have a particular value of the predictor.

Twenty-one species (22% of the total considered) were associated with tree cover (Fig. 3). Eighteen species indicated significantly variations in *Quercus* tree cover, whereas 5 species indicated variations in *Pinus* tree cover. About a half of the significant species indicated low values of *Quercus* cover, while the rest of the species indicated high values or showed optimum responses varying from 50% to 100% of *Quercus* cover. Regarding *Pinus* tree cover, only *Hedera helix* showed a progressively decreasing trend as *Pinus* tree cover increased, whereas the opposite trend was observed for *Daphne gnidium.* The rest of the species showed bell-shaped curves with optimum responses varying from 0 to 20% (*Fragaria vesca)* or from 40 to 100% (*Euphorbia amygdaloides* and *Lonicera implexa*) of *Pinus* cover.

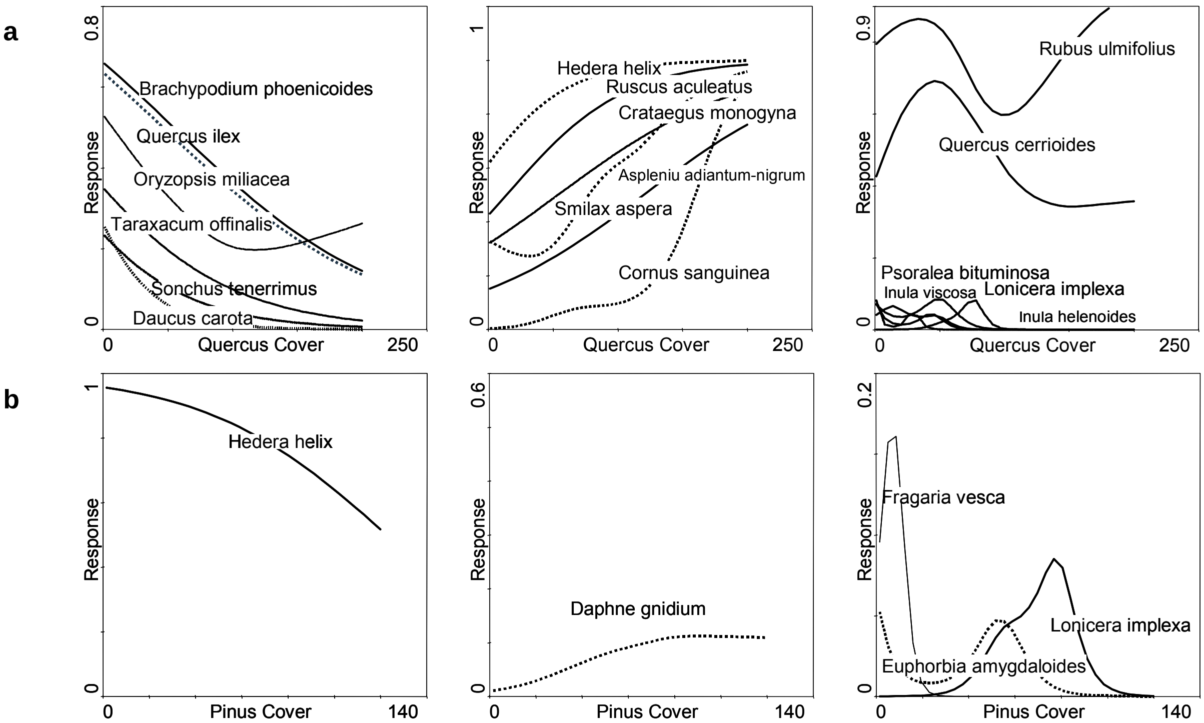

Figure 3. Significant species response ($P \leq 0.001$) for successional stage: a) *Quercus* spp. tree cover per plot; and b) *Pinus* spp. tree cover per plot. Response axes represent the probability of each species to have a particular value of the predictor.

Landscape metrics

Among the species related to landscape metrics, the majority of them 25 (26% of the total) were associated to spatial metrics (Fig. 4). Nine species were significantly related to forest patch area. For example, *Crataegus monogyna* significantly decreased and *Daphne gnidium* increased when patch area increased. Only 4 species were significantly related to mean patch fractal dimension index. *Parietaria officinalis* and *Urtica dioica* showed narrow amplitude ranges, while *Rhamnus alaternus* and *Ligustrum vulgare* were indicative of more complex forest patch shapes. Furthermore, 8 species were significantly related to forest patch connectivity: *Erica arborea, Quercus ilex, Asplenium-adiantum nigrum* and *Quercus coccifera* indicated highly connected patches, while *Celtis australis* and *Prunus domestica* were characteristic of isolated patches. The optimum response values for forest patch connectivity varied from 50 to 100 ha for *Juniperus oxycedrus* and *Arbutus unedo*. Few species were significantly related to land-use adjacent to forest patches: *Lithospermum purpurocaeruleum* and *Marrubium vulgare* were related to small perimeters of contact with shrublands and grasslands, whereas *Coriaria myrtifolia*, *Euphorbia amygdaloides, Quercus suber, Rhamnus alaternus* and *Viola sylvestris* preferred values of adjacency to crops between 30% and 70%. No species was related to road and urban area surroundings.

Twelve species were significantly sensitive to landscape metrics associated to potential human accessibility (Fig. 5). Among them, *Geranium robertianum* was mainly associated to forest edges, whereas the occurrence of *Sorbus domestica* increased with the distance to the forest edge. *Cistus salvifolius* and *Rhamnus alaternus* showed an increasing trend with increasing distance from main roads. Finally, 9 species indicated forest accessibility: 4 of them indicated highly accessible forests (*Celtis australis, Sonchus tenerrimus, Cirsium vulgare* and *Mercurialis annua*) and 5 were related to less accessible ones (*Juniperus oxycedrus*, *Phillyrea angustifolia, Erica arborea* and *Asplenium adiantum-nigrum* and *Calluna vulgaris*). No species indicated distance to unpaved roads.

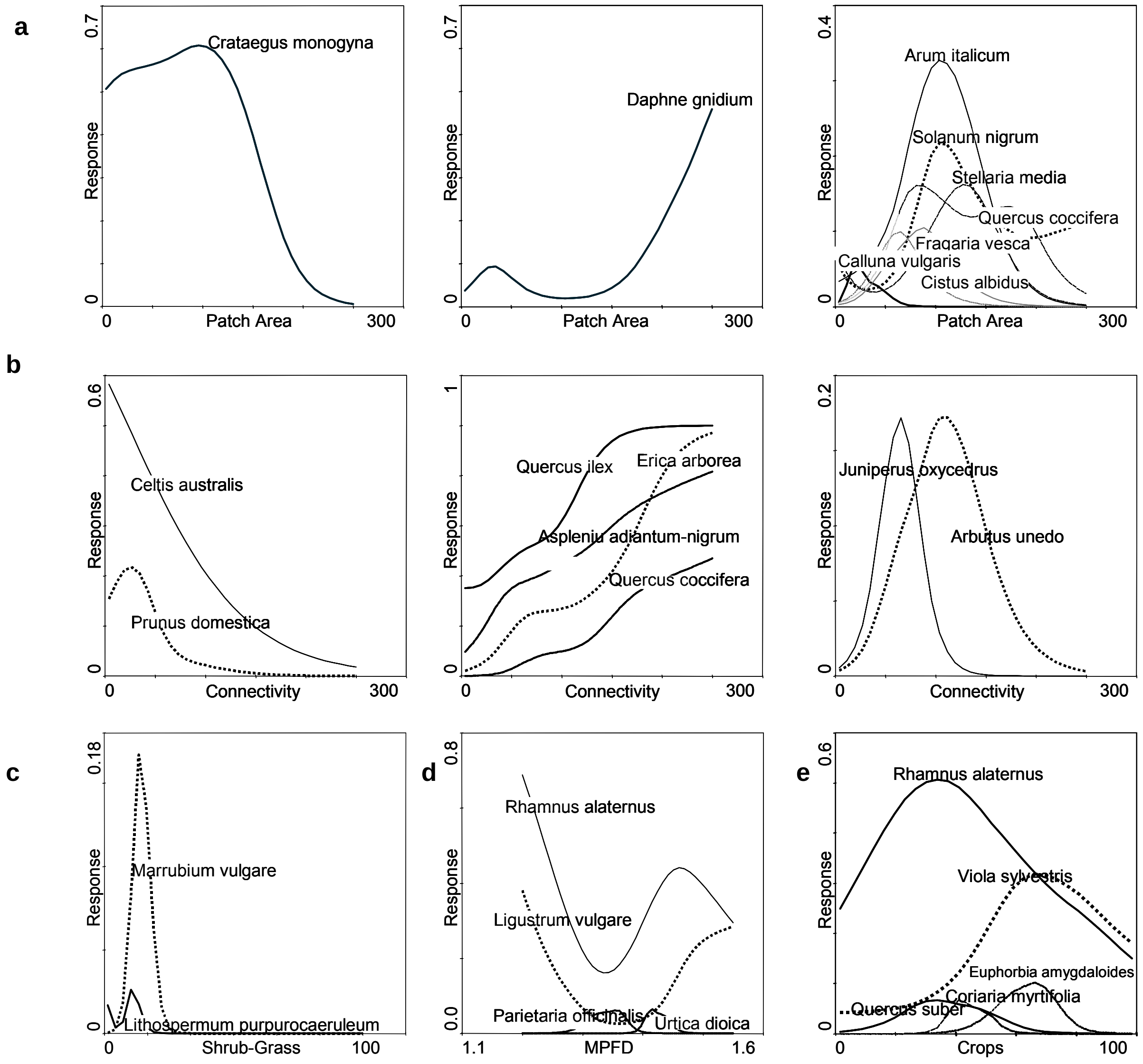

Figure 4. Significant species response (*P*≤ 0.001) for spatial metrics: a) forest patch area; b) forest connectivity; c) percentage of shrubland-grassland surrounding forests; d) mean patch fractal dimension (MPFD); and e) percentage of crops surrounding forests. Response axes represent the probability of each species to have a particular value of the predictor.

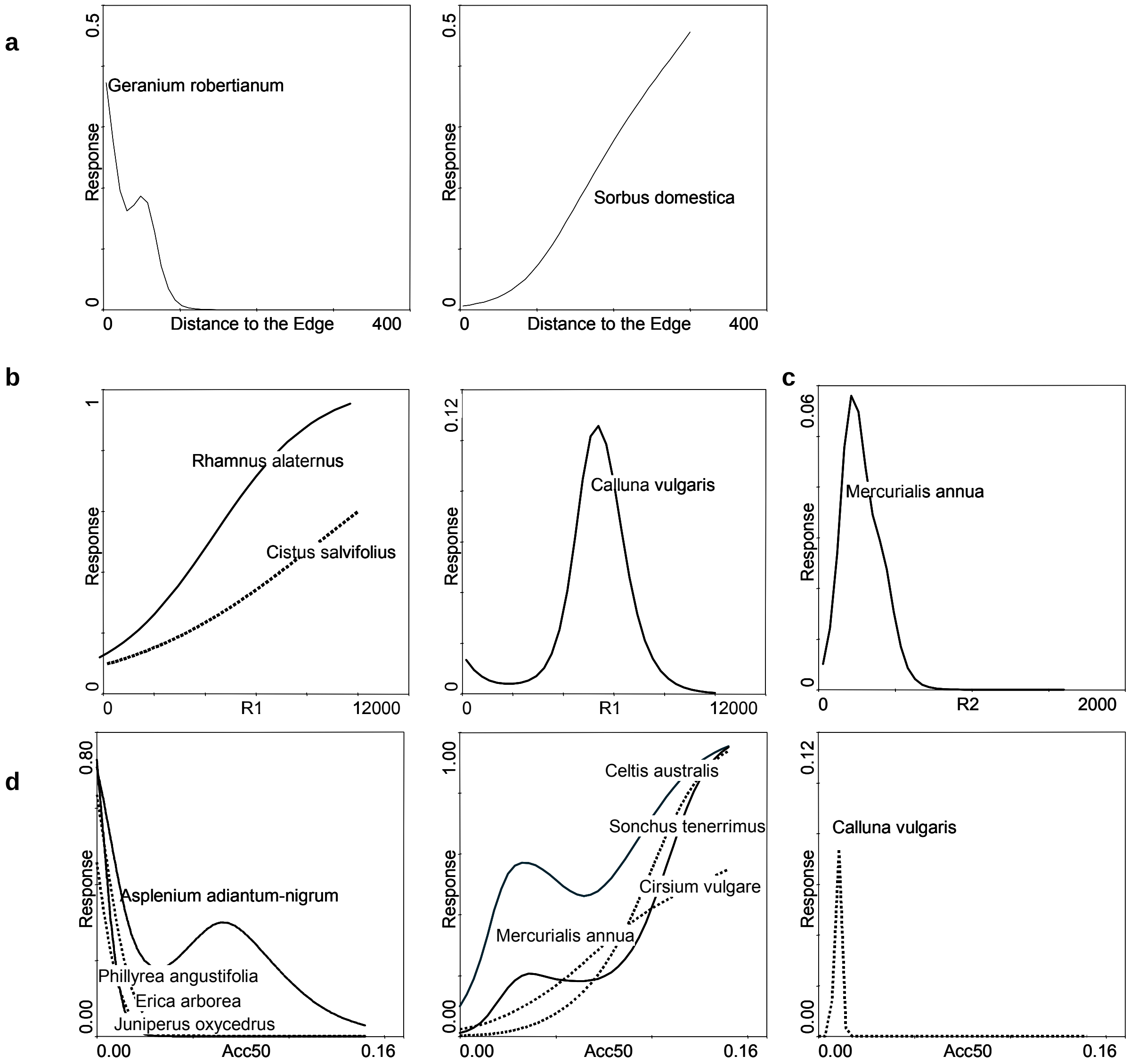

Figure 5. Significant species response (*P*≤ 0.001) for potential human accessibility variables: a) Distance to the forest edge; b) distance to the main roads; c) distance to secondary roads; and d) accessibility index. Response axes represent the probability of each species to have a particular value of the predictor.

Table 4. Generalised Additive Modelling regression results with a cubic smooth spline function and a binomial distribution with a logistic link function for all species investigated on site conditions and landscape metrics. Only values with $P \leq 0.001$ are shown. df: degrees of freedom. Species abbreviations are based on the first four letters of genus and species names (for full names, see Appendix I; For explanation of variable abbreviations, see table 1).

	UTM X		UTM Y		Slope		*Quercus*		*Pinus*		Area		MPFD		Connect.		Shrub-Grass		Crops		EdgeDist		R1		R2		Acc50	
Species	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*	*df*	*F*
Arauseri	3	6.3																										
Arbuuned			2	9.4											3	5.8												
Arumital											1	5.4																
Arundona			3	5.6																								
Aspaacut			1	17.4																								
Aspladia	2	8.3	1	30.4	1	14.8	1	12.7							3	7.4											3	5.8
Brachphoe							1	11.2																				
Buplrigi			3	11.9																								
Callvulg											3	7.4											3	6.5			3	3.7
Celtaust			1	13.7											1	14.2											3	8.5
Cirsvulg	3	6.1	3	5.4																							3	4.2
Cistalbi											1	6.2																
Cistsalv	3	5.6																					1	11.7				
Cornsang							3	5.4																				
Corimirt																			3	14.1								
Cratmono							1	14.6			1	5.5																
Daphgnid									1	11.6	1	6.3																
Dauccaro							1	24.6																				
Ericarbo	3	7.6	2	15.8											3	11.1											1	23.6
Euphamyg	3	6.4							3	7.4									3	13.2								
Fragvesc			3	8.7					2	8.7	1	6.1																
Gerarobe	3	7.9																			3	6.8						
Hedeheli							1	24.8	1	17.5																		
Inulhele	3	5.9	3	21.9	3	9.9	3	6.7																				
Inulvisc							3	10.5																				
Junioxyc			1	10.4											3	6.3											1	13.4
Lavastoe					3	10.3																						
Liguvulg													3	5.3														
Lithpurp					3	10.9											3	5.79										
Loniimpl							3	8.2	3	9.9																		
Lonixylo	3	5.6	2	11.1																								
Marrvulg																	3	18.9										
Mercannu																									3	5.8	1	7.6
Oryzmill							2	8.1																				
Parioffi			3	7.8									3	7.9														
Philangu	1	13.8	3	13.2																							1	21.4
Prundome															3	6.4												
Psorbitu							3	6.9																				
Quercerr							3	5.5																				
Quercocc											1	5.4			3	12.1												
Querilex							1	10.8							1	9.3												
Quersube																			3	8.45								
Rhamalat	2	29.7	3	8.2									3	6.1					3	8.23			1	45.0				
Rosmoffi	3	5.9																										
Rubuulmi			1	11.6			3	7.1																				
Ruscacul							1	28.1																				
Satucala			3	6.3																								
Smilaspe							3	7.5																				
Solanigr											1	5.7																
Sonctene			2	10.1			1	10.6																			3	5.8
Sorbdome			3	7.3																	1	8.0						
Stelmedi											1	5.2																
Taraoffi							1	12.0																				
Urtidioi													3	9.9														
Violsylv																			3	5.80								

DISCUSSION

Plant species composition in peri-urban forests

CCA results showed understorey species had a complex response to environmental and human-induced factors in the Vallès forest patches. This could be the consequence of heterogeneous plant species composition, but also due to the fact that individual species might be sensitive to contrasting factors. Patterns observed in this study agree with those found in other forests (e.g., Roche et al. 1998; Jacquemyn et al. 2003) indicating that both site conditions and landscape metrics play an important role in disturbed fragmented forests.

In our study, CCA revealed the existence of three species groups (i) herbaceous and woody species (most of them common *sensu* Bolós et al. 1990*)* mainly occurring in relatively old and well preserved forests showing recent fragmentation from large forest areas, and dominated by late successional stands of Q*uercus* spp.*;* (ii) woody species also forming macchias, but here associated to relatively well preserved, highly connected forest patches; and (iii) ruderal and weed species, both woody and herbaceous, indicative of new stands or highly disturbed forests, dominated by early successional stands of *Pinus* spp., highly frequented or recently managed, or potentially accessible to the urban population. These results agree with those that we obtain in chapter 3 concerning species richness of the different ecological and rarity groups, and both confirm the complexity of plant species assemblage in human-disturbed forests.

Site conditions versus Landscape

A higher number of species are related to site conditions than to landscape metrics in the Vallès lowlands, probably because site conditions directly affect understorey vegetation (Brosofske et al. 2001), whereas landscape variables have more indirect, long-term effects on species. As well as the spatial attributes summarised by patch and landscape variables, temporal patterns may also be important in determining the presence/absence of many species (White & Pickett 1985; Dale et al. 2002; Jacquemyn et al. 2003). For example the alteration of dispersal abilities due to habitat fragmentation, which could increase the time needed by some species to colonize new areas (Jacquemyn et al. 2003). Thus, we can hypothesise that understorey species respond more directly to short-term and small-scale processes than to large scale ones. Indeed, large variations in understorey species numbers have been observed following canopy disturbance by man, with a sharp increase immediately after disturbance and a decrease during the subsequent years (e.g., Gilliam 2002; Selmants & Knight 2003).

Quercus spp. tree cover was the site condition variable studied affecting most species, especially those most dependent on the presence of a thick canopy above them. This result is consistent with the results that we obtain in chapter 3, in which we also observe that *Quercus* spp. tree cover highly determine some plant species richness groups. In chapter 3, we observe that forest species richness (many considered as common by Bolós et al. 1990) increase with *Quercus* cover, whereas the opposite trend is observed for ruderal and exotic species. In the same way, Brosofske et al. (2001) found that canopy coverage was an important site variable determining plant species diversity in human-managed forests of Wisconsin (USA). This is probably due to the low light availability under the dense canopy of oak forests, which negatively affects ruderal species but is favourable to the persistence of forest plants (e.g. Deconchat & Balent 2001; Pauchard & Alaback 2004). High *Quercus* cover is also associated to low human disturbance (Guirado 2002), although the direction of the corresponding cause-effect relationship remains unclear.

Paradoxically, *Quercus ilex* seedlings exhibit a significant, negative association with *Quercus* tree cover (Fig. 3a), probably due to processes of competition and auto-inhibition (Bran et al. 1990; Bacilieri et al. 1993). In contrast, the occurrence of pine seedlings was scarce in the present study. These results suggest that a significant proportion of the new forests dominated by *Pinus* tree cover in the Vallès lowlands have been colonised by *Q. ilex* seedlings, thus illustrating a typical succession series in western Mediterranean forests.

Although many species were significantly associated to UTM X and UTM Y, it was much more difficult to extract general patterns for these variables, because of the high disparity in species responses. In chapter 3 we not find any association between plot coordinates and the richness of different ecological and rarity groups. UTM X and UTM Y may act as a surrogate of other variables which were not tested in the present study that might alter the response of some individual species in different ways (e.g., the location of cities and other urbanized areas, climatic variables such as temperature, soil type or precipitation); On the other hand, some species could be influenced by spatial autocorrelation. However, the effect of spatial autocorrelation is difficult to test when analyzing the individual responses of species, especially for those with low occurrence. Future studies could explore the effect of other new variables as well as the possibility of the autocorrelation effect to elucidate the role of forest patch location in determining individual species responses, at least for the most frequent species.

Plant species composition is considerably altered in highly man-modified areas through direct anthropogenic disturbances or by changes in landscape structure (e.g., Brosofske et al. 2001; Godefroid & Koedam 2003a, 2004b; Chapter 3). Few studies, however, have dealt with the response of individual plant species, and much less in Mediterranean peri-urban areas. We have found that site variables related to direct or potential human-induced disturbance are associated with the distribution of a large number of species in the Vallès lowlands. Increasing the accessibility of forest patches negatively affected typical Mediterranean species such as *Aplenium adiantum-nigrum, Phillyrea angustifola, Erica arborea* and *Juniperus oxycedrus*. On the other hand, accessibility promoted the presence of more ruderal and even alien species such as *Sonchus tenerrimus, Celtis australis, Cirsium vulgare* and *Mercurialis annua*. Human activity facilitates germination and propagule dispersal of these species (generally less shade-tolerant than forest ones; Deconchat & Balent 2001; Pauchard & Alaback 2004) by creating open and disturbed areas, in which light availability and competition with forest plants are respectively higher and lower than in undisturbed stands (e.g., Gilliam and Roberts 1995; Deconchat & Balent 2001; Honnay et al. 2002). In the present study, many of these species were concentrated in highly frequented and recently managed forests. These results again confirm that the forests studied, as generally expected in peri-urban areas, are highly affected by man. Indeed, the fact that only a few forest species have been identified as indicators of low disturbance suggests that the most sensitive species are very scarce, whereas the remaining species are highly resistant to such disturbances. Tellería et al. (1997) found similar results for forest bird species in Mediterranean forest archipelagos. Moreover, the increase of some ruderal species with forest accessibility (Fig. 5d) was probably related to greater human frequentation than in less accessible forests, since most accessible plots are close to urban areas and this enhances the penetration of these species across the forest edges (Pyšek 1998; Spellerberg 1998; McKinney 2002).

Surprisingly, *Celtis australis* was the best indicator of old forest management. Traditionally, this specie is characteristic of fields and crops, and is related to human activities (*sensu* Bolòs et al. 1990). However, we have observed that *Celtis australis* could be frequent in oak and cleared pine forests. In a peri-urban area like the Vallès lowlands, *Celtis australis* seeds could be dispersed from the adjacent man-modified landscape into forest by birds (Murgui & Valentín 2003), especially in open areas produced after clear-cuts. Since *Celtis australis* is a woody species, it could respond after a relatively large period of time to such forest management. Future studies might pay more attention on the response of *Celtis australis* to human disturbances in similar contexts.

The present work also highlighted that the response to habitat fragmentation can differ significantly between species. This pattern has already been pointed out for temperate forest species (Grashof-Bokdam 1997; Ehrlén & Eriksson 2000), but this is, to our knowledge, the first time that a species-specific analysis is carried out for Mediterranean species. In our study, many forest species were favoured by highly connected patches (Fig. 4b), probably because the pool of propagules able to reach forest patches increased with connectivity (Cadenasso & Pikett 2001; Pauchard & Alaback 2004). Indeed, seeds of species lacking morphological adaptations for dispersal (e.g. *Quercus coccifera, Q. ilex*) will reach distances far below those of species that do have good dispersal mechanisms (Matlack 1994). In contrast, *Celtis autralis* and *Prunus domestica* (both naturalized species) indicated isolated forests, perhaps because of high propagule pressure from adjacent human-transformed habitats or because of birds' dissemination.

In short, we have seen more species responded to site conditions than to landscape variables, probably because most of them were related to short-temporal and small-spatial processes. Moreover, a lot of the indicator species identified were related to anthropogenic disturbances, representing highly perturbed forests. Nevertheless, their indicators values were quite low, probably because we only worked with their presence/absence. We suggest that future studies might consider species coverage and abundance to improve the knowledge of their potential indicator value. Meanwhile, a general interpretation of our results enables the identification of ecological groups particularly sensitive to forest disturbances, which could be considered when developing conservation policies. Also some individual species are useful indicators of forest disturbances, physiography and landscape dynamics.

Chapter 3

Comparing the role of site condition and landscape structure on understory species richness in peri-urban Mediterranean forests

ABSTRACT

Factors affecting plant species richness in fragmented forests can act on a number of levels. Our study compares the effects of site condition and landscape structure, measured at plot- and patch-levels, on understory plant species richness per plot in peri-urban forest patches of Barcelona (NE Spain). The presence of all vascular plants was recorded on 252 plots of 100 m^2 randomly selected within 147 forest patches ranging in size from 0.25 to 218 ha. Species were divided into 6 groups, according to their ecology and conservation status. Site condition was assessed at plot level and included physical attributes, human-induced disturbance and *Quercus* spp. tree cover. Landscape structure and dynamics were assessed at patch- and plot-level from patch metrics, patch history and potential human accessibility variables. Results of multiple linear regressions indicated that the variance explained for non-forest species groups was higher than for forest species richness. Most of the variance explained was related to site conditions and landscape structure variables measured at plot level: *Quercus* tree cover was the most important factor explaining non-forest species richness per plot, which decreased when *Quercus* tree cover increased. Human-induced disturbance was an important correlate of synanthropic and total species richness per plot, which was higher in recently managed and much frequented forests. Potential human accessibility also affected the richness of most species groups. In contrast, patch-level variables played a minor role. We conclude that human influence on species richness per plot in peri-urban forests takes place on a small (plot) scale, whereas large-scale effects attributable to landscape structure and fragmentation are comparatively less important.

INTRODUCTION

Man-induced fragmentation and land use change have been identified as the most important processes affecting species richness and composition in forests around the world (Wilcove et al. 1986; Groombridge 1992; Hobbs 2000; Wood et al. 2000). All forest types are affected by area reduction, fragmentation, and degradation (Secretariat of the Convention on Biological Diversity 2002), but those of the Mediterranean biome in particular are among the most fragmented (Wade et al. 2003). Probably because of this, biodiversity studies in Mediterranean forests have become more frequent over the last few years (Blondel & Aronson 1999; Pineda et al. 2002).

Fragmentation determines changes in landscape structure through the loss of a given habitat and variations in the spatial configuration of the remaining patches (Saunders et al. 1991; Fahrig 2003). It is widely known that such processes have deleterious consequences for forest plants. In fragmented habitats, the theory of island biogeography (Mc Arthur & Wilson 1967) predicts that species number per patch depends on patch size and distance between patches: species richness tends to be higher in larger forests than in smaller ones, and higher in forest patches nearer large forest areas than in those farther away. Common explanations for these patterns include the fact that a patch area is a surrogate for habitat diversity (Usher 1979; Peterken & Game 1984; Dzwonko & Loster 1988), or that loss in an interior forest patch is more significant than loss on the forest edge when a patch area is smaller (Levenson 1981), leading to the loss of interior forest species (Grashof-Bokdam 1997; Honnay et al. 1999a). Forest fragmentation can also be seen as an independent factor influencing patch species richness through stochastic area-dependent extinctions (e.g., Jacquemyn et al. 2002), though contradictory results have been obtained in tests on this hypothesis (e.g., Boecklen & Gotelli 1984; Debinski & Holt 2000). The combination of forest patch area and isolation particularly affects interior forest species, which usually exhibit lower dispersal capacities than edge species (Matlack 1994; Honnay et al. 1999a). However, not only fragmentation processes but also other variables related to landscape dynamics (such as patch shape or the matrix influences on forest patches), can determine landscape structure and species composition (Forman 1995b).

The ecological condition or quality of forest patches can also be an important limiting factor for the occurrence of plant species in fragmented forests (Peterken & Game 1984; Ross et al. 2002; Petit el al. 2004). However, site condition can be greatly affected, in turn, by human-induced disturbances that indirectly determine forest species assemblages. In non-equilibrium human-dominated landscapes, patch size may also indirectly affect species richness through size-dependence of the intensity of patch use by humans, and through increased outside influences in smaller patches. Therefore, this could determine the patch capacity to harbour species, and modify the predictions of species-area relationships derived from island theory. This is especially likely in the Mediterranean region, where human activities have had significant effects on forest ecosystems for thousands of years (Grove 1996; Verdú et al. 2000; Gondard et al. 2001). Indeed, Mediterranean forests have suffered strong human-induced modifications of their structure (Quézel 1978; di Castri 1981), species composition (Mooney 1988) and edge processes (Brotons & Herrando 2001b), favouring the percolation of human effects into the forest core (Tellería & Santos 1999).

Comparing the effects of site condition and landscape structure on forest plant species richness and assemblage is an issue that has obvious implications for forest conservation. However, only a few recent studies have partially addressed it. For example, Petit et al. (2004) aimed to determine whether plant species richness was mostly affected by forest patch attributes or by ecological habitat conditions in two regions of Great Britain. They found that spatial characteristics were more important than ecological site conditions in highly fragmented forests. Similarly, studies carried out in forest-crop mosaics focussed either on the effects of forest ecological condition or on fragmentation variables deriving from patch attributes (e.g., Peterken & Game 1984; Dzwonko & Loster 1988, 1989, 1992; Grashof-Bokdam 1997; Honnay et al. 1999d; Jacquemyn et al. 2003). In general, these studies underestimated anthropogenic variables as possible modulators of habitat conditions and landscape structure. In contrast, studies in urban and peri-urban fragmented forests mainly analysed the effect of human-induced disturbance, directly measured or inferred from patch or landscape attributes (e.g., Hobbs 1988; Honnay et al. 1999a; Gibb & Hochuli 2002; Godefroid & Koedam 2003b), but also failed to consider anthropogenic influences on contrasting scales. The aim of the present study was to compare the effects of a set of variables related to site conditions and landscape structure measured at plot- and patch-levels on plant species richness per unit area (species richness thereafter) in peri-urban fragmented forests, with particular consideration of direct and potential human disturbances. We hypothesize that, in peri-urban fragmented forests, patch variables are of little importance compared to site conditions and landscape structure variables at plot-level, due to the high degree of human alteration characteristic of these habitats.

MATERIALS AND METHODS

The study area and plot selection procedures have been described in chapter 1.

For each sampling plot (n=252), the presence of all understory vascular plants was recorded in an area of 100 m^2. As the sampling unit was maintained constant, results should be interpreted as differences in species density (Whittaker et al. 2001), although we retain the term species richness (e.g., Ross et al. 2002; Maestre 2004).

Species were divided into groups according to their ecology and their regional rarity in relation to the flora of Catalonia (Bolòs et al. 1990), which covers the eastern part of the Iberian Peninsula: (i) total forest species (forest species thereafter), i.e. growing in dense forests with closed canopy and in their edges; (ii) rare forest species; (iii) common forest species; (iv) synanthropic species, i.e. growing in highly man-modified habitats (e.g, Sachse et al., 1990; Pyŝek et al., 2004) which include ruderal, crop and garden species, both native and alien; (v) shrubland and grassland species, i.e. mainly growing in shrubby or herbaceous communities, but also in open forests; and (vi) all species. Rarity categories were not applied for synanthropic or shrubland and grassland species, because many of the former were crop plants, garden plants, or aliens without any specific rarity status in the literature, whereas most of the latter were common species.

A set of plot- and patch-level variables were obtained for each sampling plot. Plot variables were integrated by site condition and landscape structure variables. Site condition included physical attributes (latitude, longitude, and altitude), the percentage of *Quercus* ssp. tree cover, and direct human-induced disturbance represented by human frequentation and time since the last forest management.

Landscape structure at plot level was inferred from the following variables related to potential human accessibility: (1) distance from the center of each plot to the forest edge; (2) distances to the nearest main road, secondary road and unpaved roads; and (3) Forest Accessibility Index (P_i).

Landscape structure was also inferred at patch level, by measuring metrics and history of patches containing sampling plots. We considered the following variables of patch metrics: (i) patch size (in ha); (ii) patch shape, assessed using the mean patch fractal dimension index (MPFD, Elkie et al. 1999); (iii) the percentage of major land uses surrounding the forest patch (roads, urban areas, shrublands-grasslands, and crops); connectivity measures of each forest patch: (iv) forest area within a radius of 500 m around the patch perimeter, and (v) the total area of those forests located, either wholly or partially, within this radius. Finally, patch history over the period 1956 to 1993 was included as: a) 'unchanged patches'; (b) 'recently split patches'; and (c) 'new patches'.

For a complete information of dependent and independent variables see chapters 1 and 2.

Data analysis

Linear Pearson correlation coefficients were first used to analyse the relationships between the continuous independent variables in order to reduce multicollinearity (Graham 2003) by finding out highly correlated variables. The threshold value for deciding on redundancy was set to a correlation coefficient of $|r|>0.70$. Altitude and percentage of surrounding crops were rejected because they where highly correlated with UTM Y ($|r|= 0.752$), and the percentage of urban area surrounding forest ($|r|= 0.767$), respectively. Finally, we considered 18 independent variables (Table 1).

Table 1. Dependent and independent variables used in the study[a]. Minimum and maximum values for each variable are given.

Variables	Minimum value	Maximum value	mean (± S.D)
I. Dependent variables			
Groups of species richness per plot			
Forest species	2	20	9.88 (± 3.07)
Shrubland and grassland species	0	16	3.38 (± 3.38)
Synanthropic species	0	15	2.77 (± 3.23)
Rare forest species	0	4	0.46 (± 0.70)
Common forest species	2	17	8.96 (± 2.65)
Total species	2	37	16.33 (± 6.76)
II. Independent variables			
Plot variables			
Site condition			
Altitude[*] (m)	115	485	207.00 (± 55.30)
UTM X coordinates	429620	459684	441625 (± 7455)
UTM Y coordinates	4601140	4619420	4609674 (± 3889)
Percentage of *Quercus* tree cover[b] (%)	0	165	59.00 (± 35.60)
Human frequentation (low=1, high=2)		Categorical	
Time since the last management (recent=1;		Categorical	
Landscape structure at plot level			
Distance from the edge (m)	15	283	42.00 (± 32.90)
Distance from main roads (m)	23	8555	2467.00 (± 2067.27)
Distance from secondary roads (m)	15	1500	475 .00 (± 348.68)
Distance from unpaved roads (m)	15	412	84.00 (± 72.40)
Accessibility index (non-dimensional)	0.005	0.130	0.020 (± 0.013)
Patch and landscape variables			
Patch history (patch dynamics since 1956: Large forest fragmentation=1, Patch maintenance=2, New patch=3; see text).		Categorical	
Patch metrics			
Forest patch area (ha)	0.25	218.5	35.91 (± 47.44)
Mean patch fractal dimension index (non-	1.25	1.54	1.39 (± 0.05)
Percentage of roads surrounding forests (%)	0	52.6	3.06 (± 7.31)
Percentage of urban areas surrounding forests	0	100	16.73 (± 21.67)
Percentage of shrubland-grassland surrounding	0	100	10.75 (± 14.58)
Percentage of crops surrounding forests[*] (%)	0	100	69.47 (± 26.96)
Forest area within 500 m of the patch (ha)	1.72	203.4	48.72 (± 35.43)
Total area of forests less than 500 m from the patch[c] (ha)	1.72	24501.2	9539.00 (± 11197.00)

[a] Regression analyses were performed using all the independent variables for each dependent variable.

[b] Accumulative percentage of *Quercus ilex* and *Q. cerrioides* cover.

[c] Total area of forests located, wholly or partially, within 500 m around the patch perimeter.

[*] Variables not included in the analyses to reduce coollinearity.

Multiple linear regression analyses were then used to identify independent variables related to richness of each plant species group, after testing for normality of the dependent variables with a Komolgorov-Smirnov test. Independent categorical variables (human frequentation, time since the last forest management, and patch history), were included in the regression model as dummy variables. A forward stepwise procedure was used to assess which variables should be added to the model. The significance of each independent variable was determined from the standardized partial regression coefficient.

Plant species richness could be influenced by spatial autocorrelation. In such case, data points are not independent and this contradicts the basic assumptions of statistical tests (Fortin et al. 1989; Legendre et al. 2002). We examined the autocorrelation for each plant species richness group by calculating the index of Moran (King's approach, Cliff & Ord 1981). The results showed low to moderate levels of autocorrelation for forest species richness and synanthropic species richness groups, with values between 0.12 to 0.37. The other species groups reached values <0.1. As a result, we considered acceptable the use of forward stepwise regressions in such analyses.

RESULTS

A total of 241 plant species were recorded within the 252 plots surveyed. The average number of species per plot was 16.3 (SD= 6.7), ranging from 2 to 37 species per plot. Of these 241 species, 77 were synanthropic species, 78 were shrubland and grassland species, and 67 were strict forest species (20 of them rare forest species; see Appendix II). The rest of the species (19) were only included in the total species richness group, because it was not possible to assign them to any of the above groups.

The model accounting for the highest percentage of variance corresponded to synanthropic species richness, (adjusted r^2= 0.28, $P < 0.0001$; Table 2). Plot variables in this model explained almost twice the variance accounted for patch variables. *Quercus* tree cover, ‘forest area within 500 m around the patch perimeter’ and distance to the forest edge were negatively associated with synanthropic species richness, whereas patch area and the forest accessibility index were positively associated with it. As regards to direct human-induced disturbances, recently managed and highly human-frequented forests had higher synanthropic species richness than less recently managed and less frequented ones (Fig. 1a and 1b, respectively).

Similarly, total species richness was more affected by plot-level variables than by patch variables (almost seven times more; Table 2). The model for total species richness accounted for 17.6% of total variance (adjusted r^2, $P < 0.001$). Total species richness decreased with *Quercus* tree cover, and increased with distance to the main roads, accessibility index and patch forest area. Total species richness was also higher for recently managed and highly human-frequented forests than for less recently managed and little human-frequented ones (Fig. 3a and Fig. 3b, respectively).

Only plot variables were significant in the model for shrubland-grassland species richness, and they explained 16.8% of the variance (adjusted r^2, $P = 0.001$). The model detected a main negative effect of *Quercus* tree cover and a positive effect of the distance from the main roads on shrubland-grassland species.

The regression model for rare forest species richness was not significant ($P > 0.05$), whereas those related to forest species richness and common forest species richness captured a low proportion of the total variance (adjusted $r^2 = 0.066$ and 0.090, respectively; $P < 0.05$). Only plot variables were significant for forest species richness. The model detected significant positive effects of *Quercus* tree cover on both forest and common forest species richness. Also, the distances from unpaved roads and main roads were positively correlated with forest and common forest species richness, respectively. Finally, common forest species richness increased with patch shape complexity.

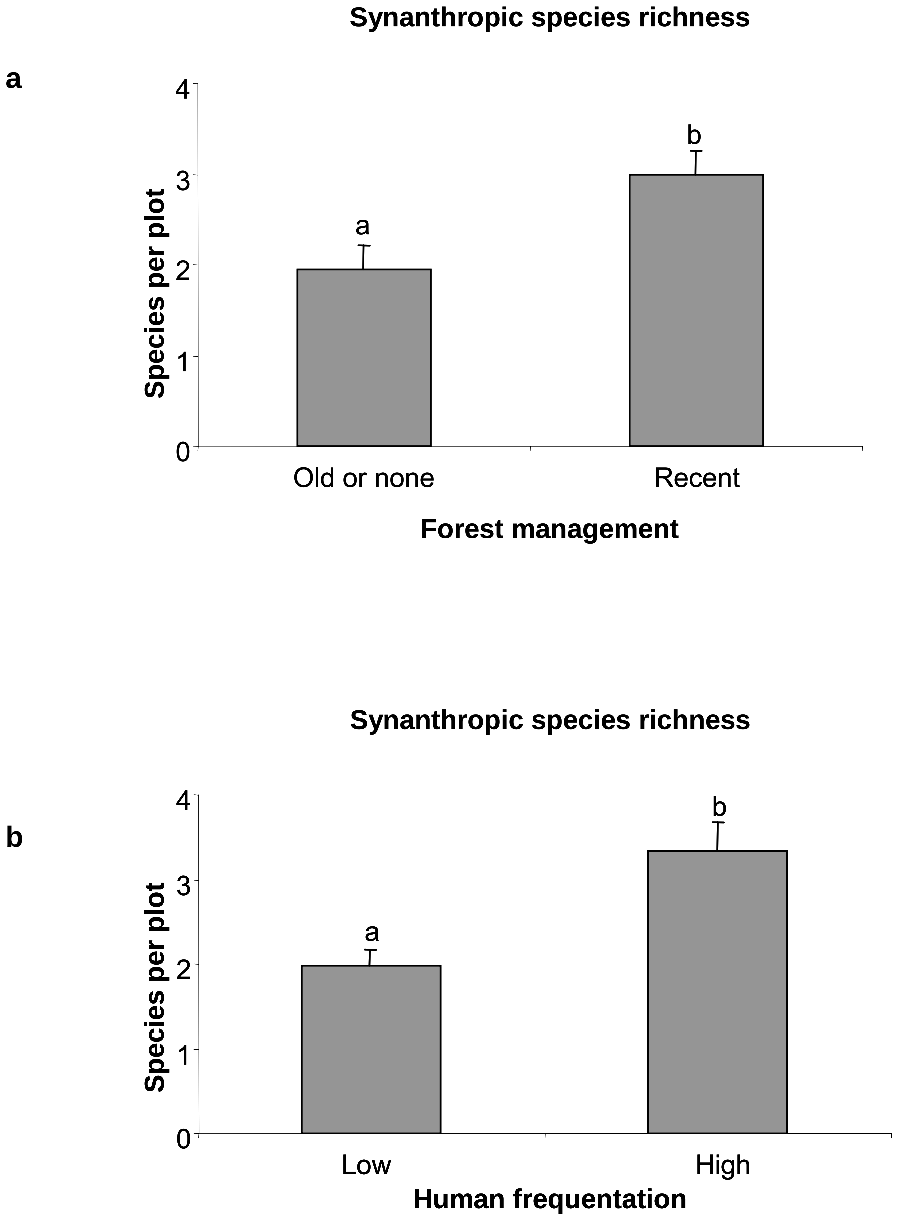

Figure 1. Significant factor effects on mean synanthropic species richness per 100 m^2 plot: (a) time since the last forest management (t=4.26; *P*<0.001); (b) human frequentation (t=2.51; *P*<0.05). Error bars are one standard error of the mean.

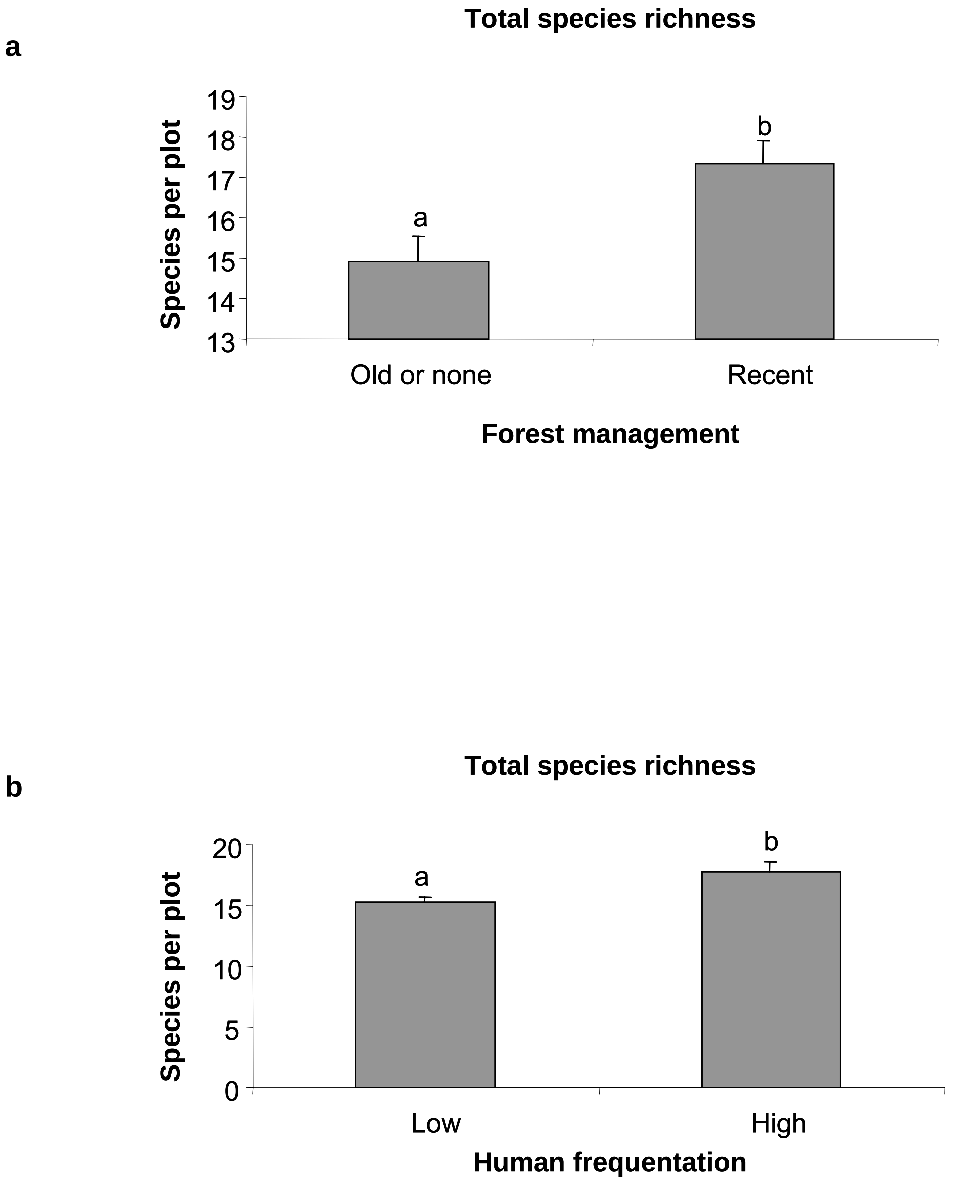

Figure 2. Significant factor effects on mean total species richness per 100 m^2 plot: (a) time since the last forest management (t=2.85; *P*<0.01); (b) human frequentation (t=2.14; *P*<0.05). Error bars are one standard error of the mean.

Table 2. Results of multiple linear regressions with a forward stepwise procedure on the plant species groups studied (beta coefficient, t test and significance). Only significant variables are shown. Significance of variables: *, *P*<0.05; ** *P*<0.01; *** *P*<0.001.

Dependent variables (adjusted r^2 of model)	Beta	t	*P*
Forest species richness			
($F_{(2,242)}=2.967$, Adjusted $r^2=0.0659$, $p<0.001$)			
Quercus tree cover	0.149	2.391	*
Distance from unpaved roads	0.131	1.984	*
Synanthropic species richness			
($F_{(12,239)}=9.043$, Adjusted $r^2=0.278$, $p<0.00001$)			
Quercus tree cover	-0.300	-5.304	***
Forest patch area	0.288	3.797	***
Time since the last management	-0.235	4.264	***
Forest area within 500 m of the patch	-0.230	-3.174	**
Accessibility index	0.215	3.848	***
Distance from the edge	-0.211	-3.418	***
Human frequentation	0.145	2.514	*
Shrubland-grassland species richness			
($F_{(7,244)}=8.259$, Adjusted $r^2=0.168$, $p<0.05$)			
Quercus tree cover	-0.300	-5.089	***
Distance to main roads	0.228	4.826	***
Common forest species richness			
($F_{(8,243)}= 4.086$, Adjusted $r^2=0.0896$, $p<0.05$)			
Mean patch fractal dimension index	0.226	3.407	***
Distance from main roads	0.210	2.721	*
Quercus tree cover	0.174	2.838	**
Total species richness			
($F_{(8,243)}=7.694$, Adjusted $r^2=0.176$, $p<0.00001$)			
Quercus tree cover	-0.237	-4.071	***
Distance from main roads	0.176	2.999	**
Time since the last management	-0.166	2.847	**
Accessibility index	0.146	2.497	*
Forest patch area	0.138	2.130	*
Human frequentation	0.131	2.144	*

For all significant models except for common forest species richness, *Quercus* tree cover was the variable with the highest contribution, while those variables having non-significant effects on species richness of the analysed plant groups were: latitude, longitude, distance from the secondary roads, surrounding land uses, patch history, and ‘total area of forests, wholly or partially, within 500 m around the patch perimeter’.

DISCUSSION

Relevance of site condition

Our results showed that site condition affected plant species richness in all species groups, confirming that these variables are important in predicting forest species richness in highly human-modified landscapes, and they need to be accounted for in landscape ecological studies. *Quercus* tree cover was the most explicative variable in the majority of the studied groups but, especially in the cases of both synanthropic and shrubland-grassland species richness (non-forest species), it decreased as *Quercus* tree cover increased. This result could arise from: (i) the lower light availability due to the dense canopy of these forests, which would negatively affect non-forest species because they are generally less shade-tolerant than forest ones (e.g. Deconchat & Balent 2001; Pauchard & Alaback 2004); and (ii) the comparatively lower human frequentation of *Quercus* forests documented in a previous work, especially affecting synanthropic species richness (Guirado 2002). It is widely known that human disturbance facilitates both seed germination from the soil seed bank and propagule dispersal of non-forest species, and that it creates open, disturbed areas, which favour non-forest species by increasing light availability and reducing competition with forest plants (e.g., Gilliam et al. 1995; Deconchat & Balent 2001; Honnay et al. 2002).

In our study, direct human-induced disturbance had a relevant association with synanthropic and total species richness, increasing both species assemblages in recently managed and in highly frequented forests. Several studies have concluded that understory species diversity increases in the first years after forest disturbance, but it can return to its original state after a few years (e.g., Gilliam 2002; Selmants & Knight 2003). Gilliam et al.

(1995) came to the same conclusion with respect to the species richness of vascular plants in Central Appalachian forests, unlike Parrotta et al. (2002), who found little impact on vascular plants species richness after harvest treatments in Amazonian forests; or Hobbs (1988), who found that native and alien species richness were negatively correlated to human disturbances in urban forest patches in Minnesota. The lack of consistency in the findings of these studies demonstrates the site-specific nature of understory responses to anthropogenic disturbances to forests (Roberts & Gilliam 1995). In our study, total and synanthropic species richness only increased in recently managed and in highly frequented forests, probably due to a set of interacting factors: (i) the increase of light availability following forest management (which favour light-demanding species); (ii) the soil disturbance favouring the germination of the soil seed bank (Brosofske et al. 2001; Deconchat & Balent 2001); (iii) the transport of propagules and (iv) the soil nutrient enrichment, both the latter two favoured by human frequentation (McDonnell & Pickett 1990; Brothers & Spingarn 1992; Parendes & Jones 2000; Moffatt et al. 2004). In short, we can conclude that direct human-induced disturbances alter natural site conditions, favouring the introduction and spread of those species related to human activities (synanthropic species), thus increasing total species richness.

Importance of landscape structure determining potential human accessibility in plots

Landscape structure and dynamics might directly affect plant species assemblage through changes in patch characteristics and configuration (Forman 1995b). They might also indirectly modify human-forest relationships, and thus would determine potential human accessibility into forest patches (Tellería & Santos 1999). In our study, we found that potential human-disturbances depending on forest accessibility can predict plant species richness. The more accessible the forest plots, the higher the synanthropic and total species richness, probably due to higher human frequentation than in less accessible forests. Also, more accessible plots are closer to urban areas and this enhances the penetration of synanthropic species across the forest edges (Pyšek 1998; Spellerberg 1998; Parendes &

Jones 2000; McKinney 2002). Indeed, the distance from the forest edge was also an important factor determining synanthropic species richness in our study, since plots nearest to the forest edge had more synanthropic species richness than those further away. The same result was found when we considered large forest areas located on the foothills adjacent to the Vallès lowlands (see chapter 5). Pauchard & Alaback (2004) found similar results when they detected a significantly higher figure for alien species richness in forest edges in comparison with interiors, but only in the case of some forest types in protected areas of south-central Chile. These results may be explained by a combination of the influence of the matrix surrounding each forest, were the influence is higher in the edges than in the interiors (Pauchard & Alaback 2004), and the fact that environmental conditions in forest edges differ from those in the interior of forest patches (Forman & Moore 1992); for example, there is lower light availability and soil pH in forest interiors (Godefroid & Koedam 2003b). Surprisingly, the distance from roads was not significant for synanthropic species richness, despite a number of empirical studies demonstrating that synanthropic (and especially alien) species increase near roads (e.g., Forman & Alexander 1998; Trombulak & Frissell 2000; Watkins et al. 2003). This may be due to the fact that we purposefully excluded the first 10 m from the forest edge, and these were probably the richest in synanthropic species (e.g., Luken et al. 1991; Watkins et al. 2003, Godefroid & Koedam 2004b).

Plot versus patch variables in peri-urban forests

As we hypothesized, the majority of the variance explained by the models was related to site condition and to landscape structure variables measured at plot level, whereas patch variables played a minor role. This might indicate the prevalence of small-scale processes over those operating at patch scale on plant species assemblage, as suggested by Honnay et al. (1999a) and Ross et al. (2002) for plant species and Gibb & Hochuli (2002) for arthropods. Patch size and isolation from other forests did not have any significant consequences on forest species richness in our study, contrary to what other authors have found (e.g., Matlak 1994; Grashof-Bokdam 1997; Honnay et al. 1999d; Petit et al. 2004). This would indicate that those forest species still persisting in the Vallès lowlands are relatively resistant to forest fragmentation, probably because most sensitive forest species disappeared long time ago, due to human disturbance spanning centuries. Similarly, Tellería & Santos (1999) observed that Mediterranean forests have been historically so heavily affected by man-induced fragmentation that interior bird species are now scarce, whereas the remaining species are able to live in the current heterogeneous landscapes.

As we expected, large patches display higher mean total species richness per plot than smaller ones probably due to forest area per se. However, forest area positively affected synanthropic species richness, contrary to expectations, and to what we found when comparing forest patches in the Vallès lowlands with large forest areas on the adjacent ranges (see chapter 5). Hobbs (1988) also found a positive species-area relationship for alien species richness in urban forest patches in Minnesota, and concluded that the species-area relationship could be altered by human disturbances within a matrix of urban land use. We suggest that such differences between studies might be due to differences in the forest size ranges under consideration. There is probably an optimal size range for forest patches, favouring the establishment of synanthropic species in the Vallès; extinction processes might increase in smaller patches, while those responsible for the introduction and spread of synanthropic species might be less active in larger forests.

Furthermore, the increase in forest patch isolation from other forests did determine higher synanthropic species richness in the Vallès lowlands, probably because of high propagule pressure from adjacent human-transformed habitats. However, we found that land use around forest patches had no significant effects on the richness of any species group, contrary to the findings of other authors (e.g. Harris 1988; Honnay et al. 1999a). Such results indicate that the edge effects in peri-urban forests are probably complex and not strictly related to the immediate human-modified habitats.

Chapter 4

Landscape and site correlates of human disturbance in peri-urban fragmented forests

ABSTRACT

Peri-urban fragmented forests are highly influenced by anthropogenic activities. However, despite the importance of direct human-induced disturbances in these forests, their territorial correlates have been poorly studied. The objective of the present study is to determine whether the main types of direct man-induced disturbance (forest frequentation and time since the last forest management) are related with site conditions and landscape structure variables in the Vallès lowlands. Generalized linear models (GLZ) with binomial distribution were constructed in order to study the relationships between the direct human-induced disturbances and the independent variables studied. Forest frequentation decreased eastwards and with increasing distance to the forest edge. No variables were related to time since last forest management. We conclude that landscape structure in peri-urban areas might be important at both regional and local scales in determining human frequentation into forest patches. We expect that in forest edges and in forests nearer the metropolitan area, direct human disturbances affect plant species richness and composition considerably, as we suggested in previous chapters. Therefore landscape structure could be a useful framework to consider for the management of peri-urban fragmented forests, to improve the conservation of these ecosystems and their organisms.

INTRODUCTION

Human modification of forest habitats is a major component of global environmental change (Vitousek 1994; Wimberly & Ohmann 2004). Previous studies have highlighted the effect of man-induced disturbances on forest composition and structure (e.g., Quézel 1978; di Castri 1981; Mooney 1988; Tellería & Santos 1999). These effects are especially evident in urban and peri-urban fragmented forests (Matlack 1993) probably due to the close contact with inhabited areas, and the high recreation pressure (e.g., Honnay et al. 1999c; Godefroid & Koedam 2003a).

Indirect anthropogenic effects on forest vegetation have been frequently associated to forest patch and landscape structure (e.g., Holt et al. 1995; Bascompte & Rodríguez 2001; Hill & Curran 2001; Kiviniemi & Eriksson 2002). Forest patch area, the distance to roads or the adjacent land use have been shown to determine the plant species richness and composition of some ecological and rarity groups in the Vallès (Chapters 2, 3 and 5), and also the cover percentage of *Pinus* and *Quercus* species (chapter 1). Direct effects of human activity on vegetation dynamics have also been considered in previous studies, especially in the case of forest management (e.g., Deconchat & Balent 2001) but more rarely for other anthropogenic disturbances (e.g., Ross et al. 2002). Among them, soil compaction associated to forest management and frequentation has been proven to determine growth reduction in some strictly forest species in the urban forests of Brussels (Belgium; Godefroid & Koedam 2004a,b). In the Vallès lowlands, total and synanthropic species richness increased in recently managed and in highly frequented forests (chapter 3), with specific taxa responding to these direct human-induced disturbances (Chapter 2). Other studies have also detected similar consequences in birds (e.g., Monkkonen & Welsh 1994) and arthropods (e.g., Gibb & Hochuli 2002).

However, despite the importance of direct human-induced disturbances in peri-urban forests, their territorial pattern and possible correlates have been poorly studied. There is some preliminary evidence that human effects on Vallès forests might be, in turn, determined by landscape and site condition variables such as the distance to the forest edge or to roads, the successional stage or the adjacent land uses. In chapter 5 we demonstrate that forests adjacent to urban areas are highly affected by high human frequentation and that, among them, stands near forest edges are more prone to disturbance than those farther away, thus asserting the importance of patch and landscape variables in determining human preferences to visit a given forest area.

The present chapter describes a preliminary study whose purpose is to determine whether the main types of direct man-induced disturbance (forest frequentation and time since the last forest management) are related to site condition and landscape structure variables in the Vallès lowlands. Such results will have important implications in the management of these forests, as well as in the better understanding of species response to these structural variables.

MATERIAL AND METHODS

The area and plot selection procedures have been described in chapter 1.

As already detected in previous chapters, two dependent variables related to direct human-induced disturbance were visually scored in each plot: evidence of human frequentation and time since the last forest management. In keeping with Gibb & Hochuli (2002), human frequentation was scored as: low frequentation, i.e. with little or no evidence of trampling and low to moderate presence of refuse; or high frequentation, i.e. when both trampling and refuse were moderate to high. The last forest management was scored as: old or non-existent management, when there were no traces of cutting or only cut stumps visible; and recent management, when we found cut branches.

A set of explicative variables were selected among forest conditions and landscape structure variables: (1) Forest patch area; (2) the distances between plots and the nearest main road, secondary road and unpaved road; (3) an accessibility value for each plot, which provides an aggregate measure of how accessible each study plot is to people; (4) the distance of each plot to the forest edge. We also included (5) *Quercus* spp. and (6) *Pinus* spp. tree cover per plot, as a measure of forest successional stage (*Quercus* spp. being late-successional compared to *Pinus* spp.). Finally, plot location (UTM X and UTM Y) was also included to detect spatial correlates on a larger scale. For a complete description of the independent variables see chapter 1.

Data analyses

Generalized linear models (GLZ) with binomial distribution and logarithmic link function were constructed in order to study what were the relationships between the direct human-induced disturbances (human frequentation and time since the last forest management) and the independent variables studied. GLZ can be used to predict responses for both dependent variables with binomial distributions and for dependent variables which are nonlinearly related to the predictors (McCullagh & Nelder 1989). Dependent variables were coded as 0 (low) and 1 (high) in the case of frequentation, and 0 (recent) and 1 (old) in the case of time since last forest management. The Wald statistic was used to ascertain the significance of each independent variable on each dependent variable. This statistic is a test of significance of the regression coefficient, and is based on the asymptotic normality property of maximum likelihood estimates. We considered as significant those variables with $P \leq 0.01$. The analyses were performed with Statistica 6.0.

RESULTS

Two variables were significantly related to human frequentation: UTM X was the most explicative variable, followed by the distance to the forest edge. The parameters estimates indicated that forest frequentation was low when the UTM X and the distance to the forest edge increased (Table 1; Fig. 1 a,b respectively). No variables were significantly related to time since the last forest management (Table 2).

Table 1. Generalized linear model with binomial distribution and logit-link function to account for variation in forest human frequentation. Significance of variables: *** *P*<0.001.

Effects	Maximum-likelihood parameter estimate	Standard error	Wald statistic	*P*-value	
UTMX	-0.0001	<0.001	15.008	<0.001	***
UTMY	0.0001	<0.001	5.946	0.013	
Patch area	-0.0015	0.004	0.187	0.662	
Distance to main roads	-0.0001	<0.001	2.401	0.119	
Distance to secondary roads	-0.0007	<0.001	2.676	0.099	
Distance to unpaved roads	-0.0050	0.002	4.604	0.026	
Accessibility index	12.6902	13.837	0.841	0.340	
Distance to the edge	-0.0221	0.007	11.259	<0.001	***
Quercus cover	-0.0072	0.005	2.254	0.132	
Pinus cover	0.0063	0.006	1.278	0.257	

Table 2. Generalized linear model with binomial distribution and logit-link function to account for variation in time since the last forest management.

Effects	Maximum-likelihood parameter estimate	Standard error	Wald statistic	*P*-value
UTMX	<-0.0001	<0.001	<0.001	0.594
UTMY	<0.0001	0.001	0.001	0.678
Patch area	-0.0050	0.003	0.003	0.110
Distance to main roads	0.0002	<0.001	<0.001	0.051
Distance to secondary roads	0.0001	<0.001	<0.001	0.740
Distance to unpaved roads	0.0015	0.002	0.002	0.453
Accessibility index	4.5340	12.313	12.313	0.712
Distance to the edge	0.0052	0.005	0.005	0.246
Quercus cover	-0.0002	0.004	0.004	0.958
Pinus cover	-0.0069	0.005	0.005	0.173

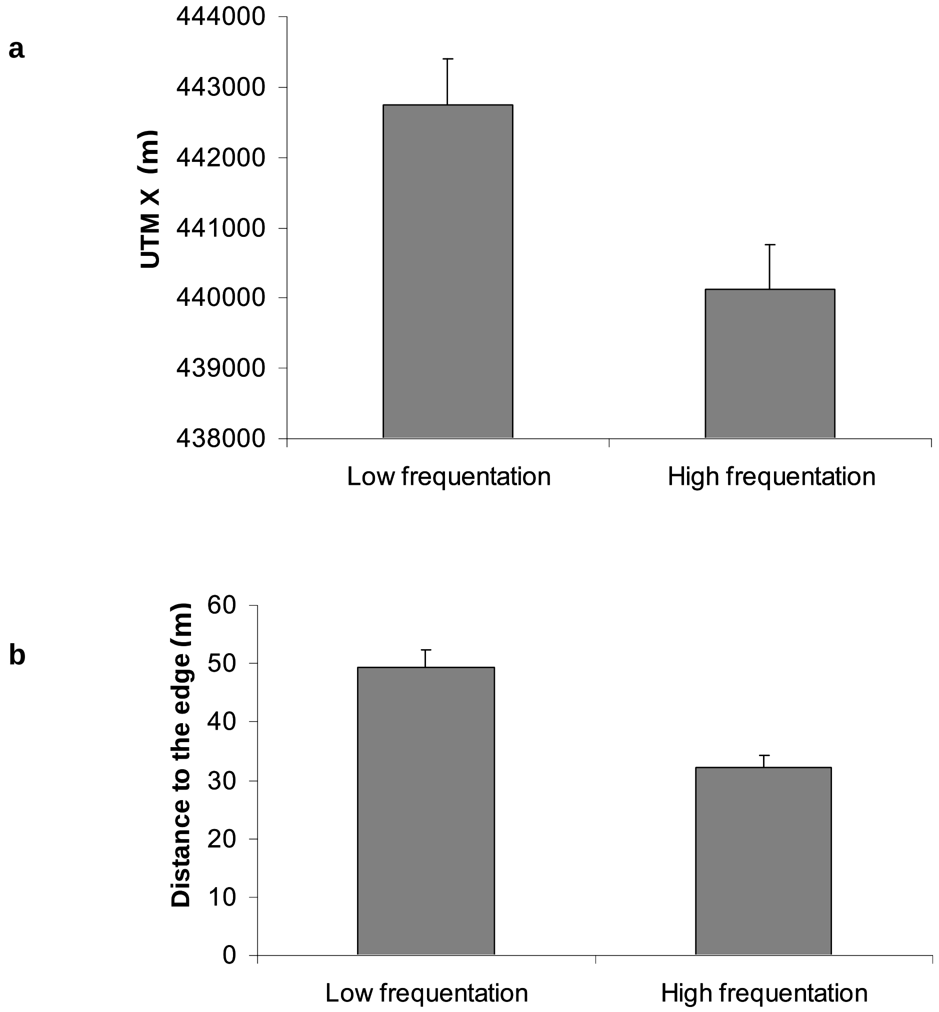

Figure 1. Significant factors mean on human frequentation categories for: (a) UTM X; and (b) distance to the forest edge. Error bars are one standard error of the mean.

DISCUSSION

The study demonstrates the existence of land correlates for human frequentation operating on two spatial scales: regional (UTM X) and local (distance to the forest edge). These factors also have significant effects on forest structure and composition (Chapters 1, 2, and 3) and a set of understory species responding significantly to them (Chapter 2). At local scale, the distance to the forest edge appeared as a significant correlate of human frequentation.

Previous studies have demonstrated that edge effects might be particularly intense as regards plant species richness and composition in forest archipelagos of urban and peri-urban areas (e.g., Matlack 1993; Ross et al. 2002; Pyšek et al. 2004). We also found that distance to the forest edge determined synanthropic species richness in forest patches of the Vallès lowlands (chapter 3), since plots nearest to the forest edge had more synanthropic species richness than those farther away. We attributed this to the ecological differences between edges and interiors (Ries et al. 2004) and to the influence of anthropogenic uses surrounding each forest, which might provide potentially intense human disturbances (e.g., Matlack 1993; Ross et al. 2002; Pauchard & Alaback 2004). The significant relationship between human frequentation and the distance to the forest edge in the present study contribute to support that, in peri-urban areas, anthropogenic influences might be as important as structural and environmental factors. The same result was found when considering a sub-set of lowland forest patches in combination with large forest areas of the adjacent foothills (chapter 5), thus supporting this hypothesis.

The significance of UTM X in the present study reveals that forest plot location is also important in explaining human frequentation at a large scale, possibly because it is a surrogate of other variables, like for example major density of cities or of roads in some sectors of the study site. Forman et al. (2003) suggested that the additive effect of roads, urban areas and others might be important at a large scale. In the study area, the South-Eastern zones present high densities of human land uses which might determine different levels of human frequentation into forests. We suggest that the increasing human disturbance westwards in the Vallès lowlands is due to the increasing influence of the conurbation.

Distance to unpaved roads is a key element of landscape configuration in man-modified areas, despite the fact that it was only marginally correlated with human frequentation in the present study. Roads are landscape elements that contribute to forest fragmentation and forest removal processes (Reed et al. 1996; Saunders et al. 2002; Watkins et al. 2003), and that have shaped the way modern humans interact with forests, providing people with more opportunities to venture into them (Trombulak & Frissell 2000; Watkins et al. 2003). Unpaved roads in particular are important elements determining human visit into forests, since most of them were built to promote timber harvest and cross forest patches (Miller et al. 1996; Trombulak & Frissell 2000). Actually, unpaved roads are widely used for hikers, riders, bikers, for off-road vehicles or for people that go there to dump rubbish. Such impact results in major and persistent adverse effects on the native flora and fauna (Seibert 1993). Godefroid & Koedam (2004b) found that forest paths in the urban forests of Brussels (Belgium) have a significant effect on soil compaction, generally increasing in the vicinity of paths, and attributed to visitors' trampling. In turn, such soil compaction had important consequences on plant species assemblage, generally increasing the amount of ruderal species (Godefroid & Koedam 2004a,b).

Forest patch area was not significantly associated to human frequentation in the present study. Previous works suggested that human visitation could particularly favour the percolation of synanthropic species and the decrease of native species in small forest patches, because they have high perimeter-area ratios and may experience higher densities of visitors per unit area than larger patches (e.g., Laurance & Yensen 1991; Hill & Curran 2001; Ross et al. 2002). However, the lack of significance in the present study and in a previous work including a sub-set of the Vallès forest patches (chapter 5) suggests that forest size could be a secondary variable in highly man-modified contexts. We also suggest that it might be important the range of forest patch sizes considered.

In contrast to human frequentation, structural variables were not related to the time since last forest management, probably because the effects resulting from the specific use an owner makes of his forest override the effects of these variables (Tucker 1999), or because the estimate of forest management considered in the present study can only detect changes in the short term.

In short, landscape structure in peri-urban areas is a significant correlate of the relationship between forests and human frequentation into them. Considering the results obtained in the present chapter and in the previous ones, we can expect that forest edges in the Vallès, and in general near metropolitan areas, are subject to a gradual extinction of rare forest species and the invasion by ruderal species, both native and alien. This will lead to an increasing homogenisation of forest flora in metropolitan regions, with profound effects on the conservation of plant biodiversity at regional scale (McKinney & Lookwood 1999). Considering landscape structure variables to interpret human frequentation into forest plots could provide a useful framework for the management of peri-urban fragmented forests, to improve the conservation of these ecosystems and their organisms.

Chapter 5

Understory plant species richness and composition in metropolitan forest archipelagos: Effects of forest size, adjacent land use and distance to the edge

ABSTRACT

The aim of the present study is to address the relative role of adjacent land use, distance to forest edge, forest size and their interactions on understory plant species richness and composition in perimetropolitan forests of Barcelona (Catalonia, NE Spain). Twenty sampling sites were distributed in two forest size-categories: small forest patches (8-90 ha) and large forest areas (>18000 ha). For each forest-size category, 5 sites were placed adjacent to crops, and five sites adjacent to urban areas. Vascular plant species were recorded and human frequentation was scored visually in 210 10x10 m plots placed at 10, 50 and 100 m from the forest edge, and additionally at 500 m in large forest areas. Plant species were grouped according to their ecology and rarity categories. A Non-metric Multidimensional Scaling (NMS) ordination was carried out to detect patterns of variation in species assemblage, and to explore the relationships between these patterns and the richness of the species groups and the studied factors. Factorial ANOVAs were used to test the significance of the studied factors on the richness of species groups. Relationships between human frequentation and the studied variables were assessed through contingency tables. Forest-size category was the main factor affecting synanthropic species (i.e. those thriving in man-made or man-disturbed habitats). Synanthropic species richness decreased with increasing distance from the forest edge and, when forests were adjacent to crops, it was higher in small forest patches than in large forest areas. Richness of rare forest species was lower in small forest patches than in large forest areas when forests were adjacent to urban areas. Richness of common forest species and of all forest species together were higher close to the forest edge than far from it when forests were adjacent to urban areas. Forests adjacent to urban areas were more likely to experience high human frequentation, particularly in those plots nearest to the forest edge. Forest-size category and adjacent land use were the most important factors determining species richness and composition. The preservation of large forests adjacent to crops in peri-urban areas is recommended, because they are less frequented by humans, are better buffered against the percolation of non-forest species and could favor the persistence of rare forest species.

INTRODUCTION

Forests in the Mediterranean region have suffered great changes in their extent, structure and composition due to the land-use changes produced over millennia (Thirgood 1981; Grove & Rackham 2001). Two processes associated with land-use change are habitat loss and fragmentation, which ultimately lead to a reduction in forest species richness by increasing the extinction risk of local populations and reducing colonization of isolated fragments (Holt et al. 1995; Bascompte & Rodríguez 2001; Hill & Curran 2001; Kiviniemi & Eriksson 2002; Godefroid & Koedam 2003b). Forest fragments also become susceptible to invasion by non-forest species coming from the surrounding man-modified habitats (Murcia 1995; Morgan 1998). Extinctions of the most vulnerable plant species and invasion by non-forest species seem to be especially intense in fragmented peri-urban forests (Honnay et al. 1999c; Hobbs 1988, 2000; Godefroid & Koedam 2003b).

Forest fragmentation exposes the organisms that remain in the fragment to 'edge effects', the conditions of a different surrounding ecosystem (Murcia 1995; Cadenasso & Pickett 2001). Since forest edges are the first point of contact for seed flux between a forest fragment and its adjacent habitats, the percolation of non-forest species can occur first across them (e.g., Saunders et al. 1991; Brother & Spingarn 1992; Wiens 1992; Forman 1995b; Honnay et al. 1999a; Cadenasso & Pickett 2001; Hersperger & Forman 2003). Many studies have dealt with the issue of non-forest species advancing into fragmented forests from their edges (e.g., Palik & Murphy 1990; Cadenasso & Pickett 2001; Pauchard & Alaback 2004), but with little consensus in the results (Murcia 1995; Ries et al. 2004). Species richness and composition in forest edges are not only affected by physical changes in relation to the proximity to the edge (e.g., light availability, wind influences), but also by propagule pressure that is mainly determined by the nature of adjacent land uses (Cadenasso & Pickett 2001). Despite its potential importance for forest conservation, the effect of adjacent human land uses on forests has only been partially explored, with most studies focusing on roads (e.g., Reed et al. 1996; Spellerberg 1998; Forman et al. 2000; Parendes & Jones 2000; Forman et al. 2003; Pauchard & Alaback 2004) and, less frequently, on croplands (e.g. Fraves 1994; Honnay et al. 1999a). The effects of urban areas on the species composition of adjacent forest patches have received less attention (Honnay et al. 1999c), despite the high recreation pressures that those patches experience (Matlack 1993; Godefroid & Koedam 2003b). Such pressure can favor the flow of propagules of ruderal and garden plants, both native and exotic (Brothers & Spingarn 1992; Hobbs 2000), which in turn can affect the persistence of rare understory species (Honnay et al. 1999a).

As Murcia (1995) indicated a decade ago, there are few studies addressing the interaction between the different factors determining edge effects. Some recent works (e.g., Cadenasso & Pickett 2001; Godefroid & Koedam 2003a; Pauchard & Alaback 2004) has shown non-linear responses of species richness and composition to the distance from the edge. Patch size is obviously important since the proportion of edges in relation to core area increases in small patches, thus enhancing edge-related processes (e.g., Forman & Gordon 1986; Santos & Tellería 1992). Edge effects on forest species richness and composition are also expected to be particularly intense in forest archipelagos of peri-urban areas, because the surrounding man-modified landscape matrix provides potentially intense human disturbances that can modify the effects of fragmentation (Matlack 1993). The present paper is aimed at addressing the relative role of forest size, adjacent land use (crops and urban areas), distance to the forest edge, and their interactions on understory species assemblage. With this purpose, we tested the effect of these factors on the number of total species per plot and on the number of species belonging to the main ecological and rarity groups.

MATERIALS AND METHODS

Study site

The present study was conducted in the Vallès lowlands and adjacent foothills, in the northern metropolitan area of Barcelona (Catalonia, NE Spain; Fig. 1). This area is particularly suitable for analyzing urban and agricultural effects on forest patches since it contains a complex mosaic of urban areas, croplands, and forests of variable size. The climate is typically Mediterranean, characterized by mild winters and dry and hot summers, with a mean annual precipitation of 600-700 mm. Potential vegetation is constituted by broadleaved forests of *Quercus ilex* and *Quercus cerrioides*, which have been mainly eliminated in favor of crops or urban areas, or substituted by mixed forests of *Quercus* spp. and secondary Mediterranean pines (*Pinus halepensis*, *P. pinea*, *P. nigra*). The lowland landscape is a complex mosaic of croplands (53% of the area; CREAF-DMA, 1993), forests (31%), urban and industrial areas (12%) and transport infrastructure (3%), and others (1%). Lowland forests are organized in multiple patches of small to medium size (up to 250 ha) and one of 490 ha. In contrast, more homogeneous landscapes made up of large forest areas of hundreds or thousands of hectares dominate the adjacent foothills (Gracia et al. 2000). Forests of lowlands and their adjacent foothills show little difference in their dominant plant communities.

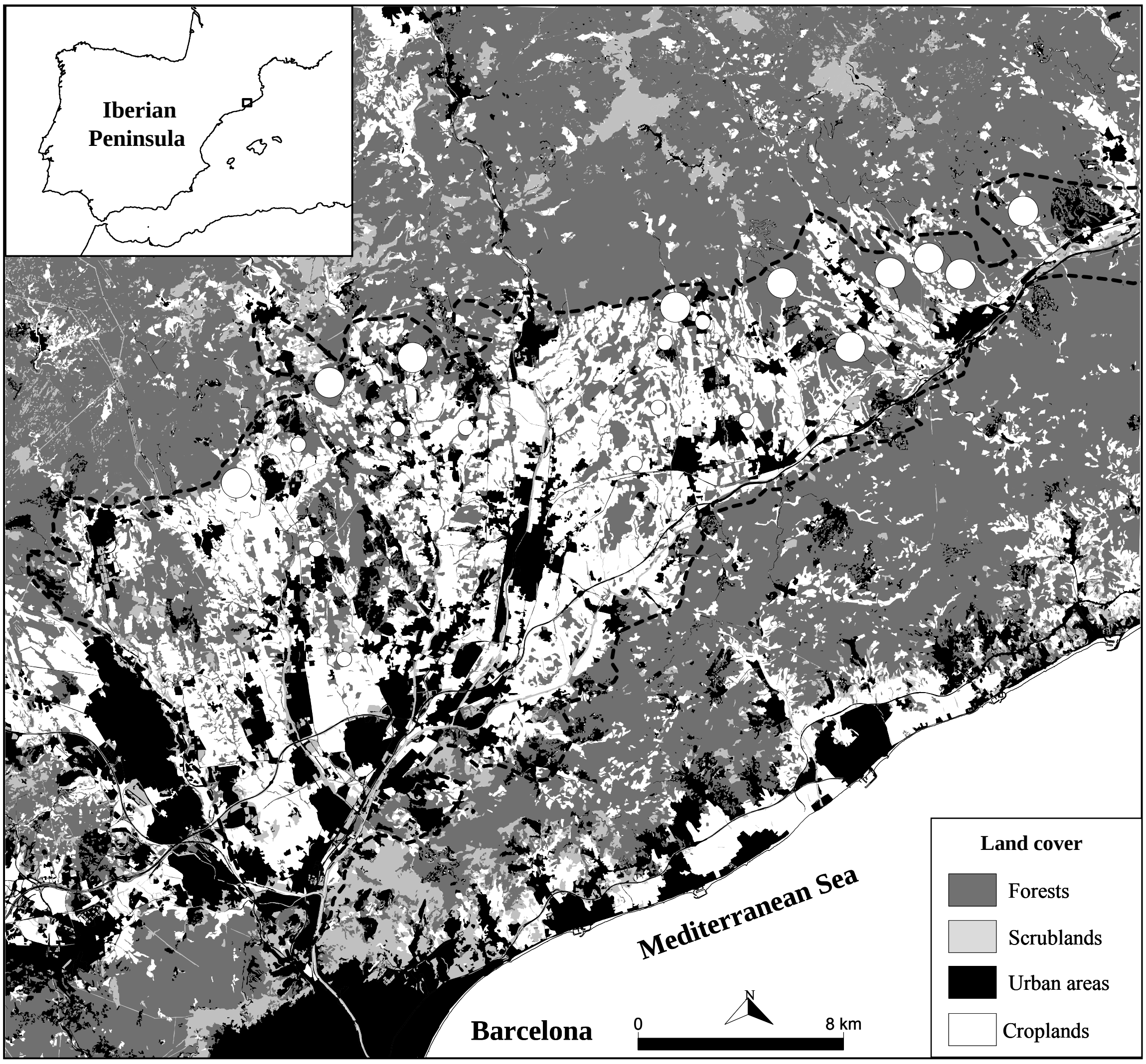

Figure 1. Location of the study area in the northern sector of the Barcelona conurbation. Large circles indicate large forest areas, whereas small circles indicate small forest patches. The area inside the dashed line indicate slopes < 20º.

Sites selection

We selected 20 sampling sites distributed in two forest-size categories: small forest patches (ranging from 8 to 90 ha) and large forest areas (> 18000 ha) in areas with slope <20º and elevation ranging from 115 to 500 m a.s.l.. A single sampling site was located in each of 10 small forest patches. Seven additional sampling sites were located in one large forest area, and 3 sites in a second large forest patch. The small forest patches are 1 to 4 km from the nearest large forest area. For each forest-size category, 5 sites were placed adjacent to crops, and the other 5 adjacent to urban areas.

A first pre-selection of sites was performed using 1:25000 orthophotomaps, generated by the Cartographic Institute of Catalonia in 2000. We selected sites in forests with (1) as regular (isodiametric) form as possible; (2) at least 400 m of contact perimeter with the selected adjacent land use (urban or agricultural); and (3) a minimum diameter > 200 m for small forest patches and >1000 m for large forest areas. Among these patches, we selected forest stands with a mixed canopy of *Quercus ilex* (mean *Quercus* tree cover= 52.1%; s.e. 1.9) and *Pinus* spp. (*Pinus halepensis* or *P. pinea;* mean *Pinus* tree cover = 25.0%; s.e. 1.6), the dominant forest type in the study area.

At each site, we set up three replicated transects from the edge to the interior of the forest. In small forest patches, we marked three 10x10 m plots per transect at 10, 50 and 100 m from the edge towards the center of the forest, whereas in large forest areas we put four 10x10 m plots per transect at 10, 50, 100, and 500 m from the edge (Fig. 2). We have purposefully excluded the narrow outer belt (0-10 m) of each forest patch where edge effects are trivially stronger. Patch shape and transect set up ensured that, for all plots, the nearest forest edge was the edge where the respective transect started. In total, we surveyed 90 plots for small forest patches and 120 plots for large forest areas.

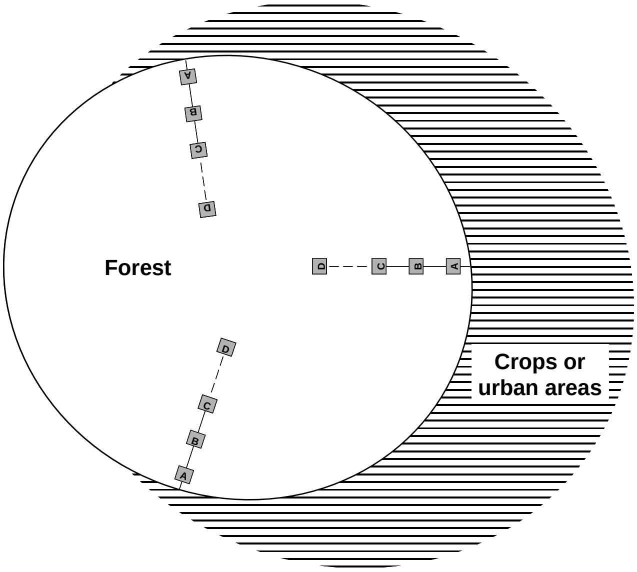

Figure 2. Diagram of plot and transect layout used to sample small forest patches and large forest areas on 20 sites in urban and peri-urban forests of the metropolitan area of Barcelona. The distance of each plot to the forest edge was A: 10 m; B: 50 m; C: 100 m; and D: 500 m (the last one was present only in large forest areas).

Most non-experimental studies like ours could be affected by pseudoreplication, which might inflate Type I error in the statistical analysis. To minimize sources of pseudoreplication, sites of the same combination of forest-size category and adjacent land use were set up along the 40 km of the main axis (NE-SW) of the territory, with each of them being located as far as possible from the others. A previous study in the Vallès lowlands (see chapter 3) found no effects of latitude and longitude on the species richness of ecological and rarity groups within our study area. Also, no significant differences were found for *Quercus* tree cover and *Pinus* tree cover between small forest patches and large forest areas (ANOVA; F= 2.85 and *P*=0.092 for *Quercus* tree cover; F= 0.002 and *P*=0.99 for *Pinus* tree cover). Both these results highlight the general uniformity of the forest flora over our study area. Pseudoreplication was further decreased through pooling data before ANOVAs were carried out, as explained below.

Edge orientation can be a potential cause of uncontrolled variation because it determines microclimatic differences (e.g., Didham & Lawton 1999; Norton 2002). Our study does not deal with edge orientation effects, and to filter out such effects, the three replicated transects of each sampling site in small forest patches were orientated in different directions. We also ensured that sites in large forest areas covered a wide range of edge orientations for each adjacent land use.

Vegetation sampling

A list of all vascular plant species was compiled for each plot during the springs of 2002 and 2003. The species were grouped according to their ecology following Bolòs et al. (1990): (i) strict forest species (forest species thereafter; growing in closed-canopy forests and their edges); (ii) shrubland and grassland species (mainly growing in scrubby or herbaceous communities, but also in open-canopy forests); (iii) synanthropic species (vegetation growing in man-made or man-disturbed habitats; e.g., Sachse et al., 1990; Pyŝek et al., 2004), which include ruderal, crop and garden species, both native and alien; and (iv) all species. Also the rarity category in the eastern Iberian Peninsula was obtained for each forest species from Bolòs et al. (1990), separating (v) common forest species; and (vi) rare forest species (see Appendix II). Rarity categories were not applied for synanthropic species, because many of them are crop plants, garden plants, or aliens without specific rarity status in the literature. Shrubland and grassland species were not scored for rarity since most of them are common species. Indirect evidence for the intensity of human frequentation was visually scored in the field as either: (i) low, i.e. with little or no evidences of trampling or rubbish; or (ii) high, i.e. with considerable or high presence of either trampling or rubbish.

Data analysis

An ordination of sampling plots based on their understory species composition (presence/absence) was performed to detect the patterns of variation in species assemblages, and to explore the relationships between these patterns and the richness of the species groups and the studied factors (distance to the forest edge, land use adjacent to the forests, and forest-size category). The ordination was done using Non-metric Multidimensional Scaling (NMS), a particularly well suited method for the analysis of floristic data because no assumptions are made concerning the data, thus providing a robust ordination (McCune & Mefford 1999). We ran the NMS with all plots excluding the 500-m plots because they were only sampled in large forest areas. Only those species detected in more than one plot (n=140) were used in the ordination. A varimax rotation was applied to maximize the loading of individual variables on the dimensions of the reduced ordination space. Finally, we used Pearson's linear correlation coefficients to explore the association between the main ordination axes and the richness of the plant species groups.

Factorial ANOVAs were used to test the effects of distance to the forest edge, adjacent land use, forest-size category (small patches *versus* large forest areas) and their interactions on species richness of the different ecological and rarity groups. To avoid pseudoreplication as much as possible, we averaged species richness over the three plots located at the same distance from the edge for each sampling site and used the resulting mean as input for the ANOVA. Each set of three replicated plots was thus treated as a single sampling unit. Two separate ANOVAs were performed: (a) considering the three above mentioned factors and thus excluding the 500-m plots of large forest areas in order to compare the edge effects between small patches and large forests; and (b) considering adjacent land-use and edge distance for large forest areas only, thus including the 500-m plots, to study the changes in understory richness farther away from the edge.

All dependent variables were log-transformed to improve. Pairwise comparisons were performed for the significant factors and interactions using the LSD test. Finally, contingency tables were constructed to evaluate the potential association between human frequentation and the studied factors (distance to the forest edge, land use adjacent to the forests, and forest-size category). Analyses were performed for (a) all forests, thus including plots located at 10 to 100 m from the forest edge, and (b) only for large forest areas, thus including plots at 500 m. The relationship between human frequentation and distance to the forest edge was tested separately for each adjacent land use (crops and urban areas). Unless otherwise stated, all effects and interactions commented upon in the results section are statistically significant ($P<0.05$).

RESULTS

Species composition by groups

A total of 203 understory plant species were recorded, with 146 species being present in more than one plot. Of all species, 77 were forest species (27 of them rare), 75 shrubland and grassland species, and 51 synanthropic species (see appendix II). Regarding the adjacent land use, 77.5% of all species were found in forests adjacent to crops, and 78.9% in forests adjacent to urban areas. Regarding forest-size category, 75.0% of the 196 species located at the first 100 m from the forest edge grew in large forest areas (the 500-m plots were here excluded to provide for equal sample sizes among forest-size categories), while 70.9% grew in small forest patches. As for distance from the forest edge, 164 species were recorded in the plots closest to the edge (10 m), amounting to 83.7% of the species found within 100 m from the forest edge; 139 (70.9%) were recorded in the 50-m plots, and 114 (58.2%) in the 100-m plots. This percentage was not computed for 500-m plots because they were sampled only in large forest areas.

Ordination results

The best solution for the NMS ordination was made up by three significant axes (Monte Carlo test, $P<0.05$) with a cumulative r^2 of 0.738. Table 1 lists the correlations between the extracted axes and the species richness of the ecological and rarity groups. Axis 1 was closely correlated with shrubland and grassland species richness, whereas axis 2 was most correlated (negatively) with synanthropic species richness and axis 3 with forest species richness.

The distribution of forest-size categories, adjacent land use and distance to the forest edge on the NMS space (Fig. 3) shows that forest-size category was the factor that best visually grouped the plots. Small forest patches were mainly grouped in the lower part of axis 2, which is negatively correlated with synanthropic species richness. Some plots in forests adjacent to crops were concentrated on the left side of axis 1, but the overlap with plots in forests adjacent to urban areas was high. Distance to the forest edge gave no visual groups for any axis.

Table 1. Explained variance of the first three NMS ordination axes and Pearson correlation coefficients between these axes and species richness of the different species groups. Significant correlations (*P*<0.05) are indicated by an asterisk.

	Axis 1	Axis 2	Axis 3
% Explained variance	32.6	27.7	13.6
Species group			
Forest	-0.09	-0.33*	0.49*
Shrubland-grassland	0.81*	0.04	-0.10
Synanthropic	0.07	-0.61*	0.04
All species	0.45*	-0.38*	0.24*
Rare forest	-0.05	-0.04	0.42*
Common forest	-0.08	-0.36*	0.41*

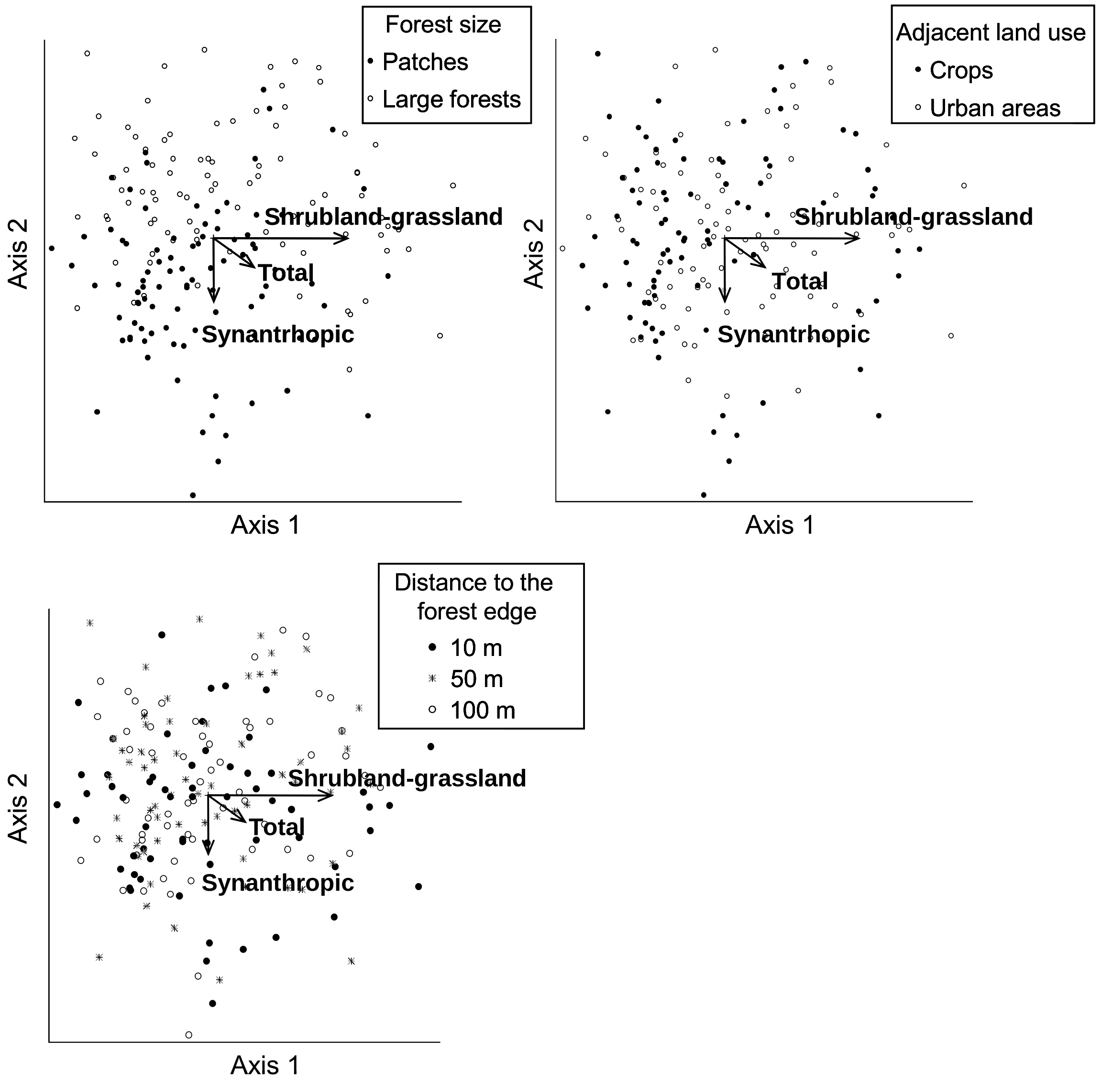

Figure 3. Ordination plot of vegetation data in the studied forests based on a Non-metric Multidimensional Scaling (NMS). Plots are grouped according to forest-size category, distance from the forest edge and adjacent land use. Length and direction of the vectors shown represent the correlations between the richness of the species groups and each axis (r^2> 0.2; Table 1).

Effects of forest-size category, adjacent land use, and distance to forest edge on richness

ANOVA results for plots between 10 m to 100 m from the forest edge are summarized in Table 2. Synanthropic species richness was higher in small forest patches than in large forest areas, whereas shrubland-grassland species richness showed the opposite pattern (Fig. 4a and 4b). Also, synanthropic species richness decreased with increasing distance from the forest edge (Fig. 4c). Adjacent land use was not significant *per se* for any species group. However, the interaction of this factor with forest-size category was significant for synanthropic species richness, which was lower in large forest areas adjacent to crops than in the other three combinations (Table 2, Fig. 4d). None of the studied factors had significant effects on total and forest species richness.

Richness of rare forest species was higher in large forest areas than in small forest patches (Table 2, Fig. 5a), but this pattern was affected by the type of land use adjacent to forests (Fig. 5c): small forest patches showed lower species richness than large forest areas when the adjacent land use was urban but not when it was crops. Common forest species richness was lower in forests adjacent to crops than in those adjacent to urban areas (Fig. 5b). The rest of the factors and interactions were not significant.

Total and forest species richness decreased significantly as distance to the edge increased when the ANOVAs were limited to large forest areas, thus including plots from 10 to 500 m from the forest edge (Fig. 6a; Table 3). For forest species, richness decreased more quickly when the adjacent area was urban rather than cropland (Fig. 6b). No significant effects of the studied factors were found for synanthropic nor shrubland-grassland species richness when the analysis was restricted to large forest areas.

Richness of common forest species decreased as distance to the edge increased when the ANOVA was limited to large forest areas (Table 3, Fig. 6c), but, as in the case of total forest species, this was related to the adjacent land use (Fig. 6d): richness decreased only in forests adjacent to urban areas, with the 10 m and the 100 m plots differing significantly from the 500 m plots.

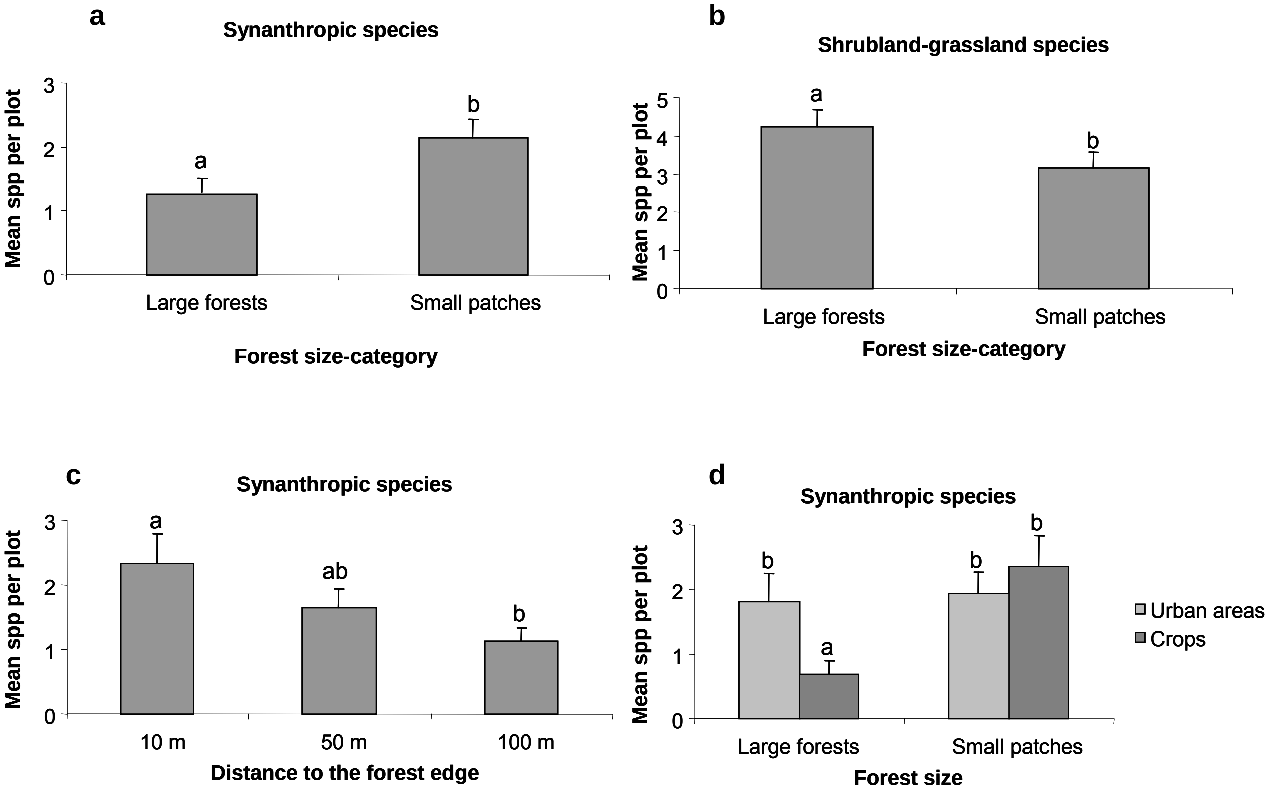

Figure 4. Significant factor effects on mean richness per plot (100 m^2) of non-forest species: forest-size category for (a) synanthropic species and (b) shrubland-grassland species, and (c) distance to the forest edge for synanthropic species. The interaction between forest-size category and adjacent land use for synanthropic species richness is shown in (d). Error bars are one standard error of the mean. Different letters show that means are significantly different (LSD test, $P < 0.05$).

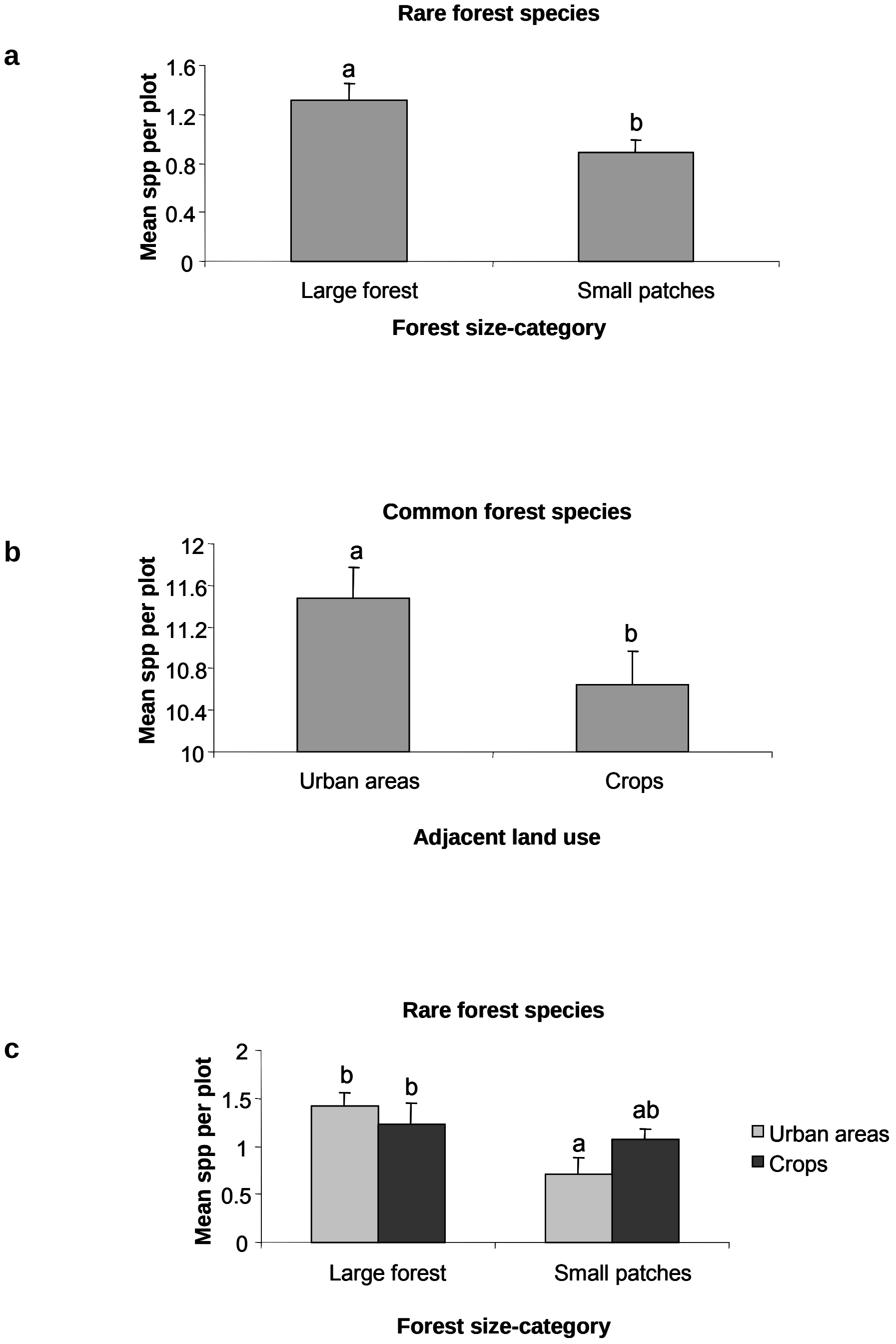

Figure 5. Forest species: Significant factor effects on mean richness per plot (100 m^2): (a) Forest-size category and rare forest species; (b) adjacent land use and common forest species; and (c) interaction between forest-size category and adjacent land use for rare forest species. Error bars are one standard error of the mean. Different letters show that means are significantly different (LSD test, $P < 0.05$).

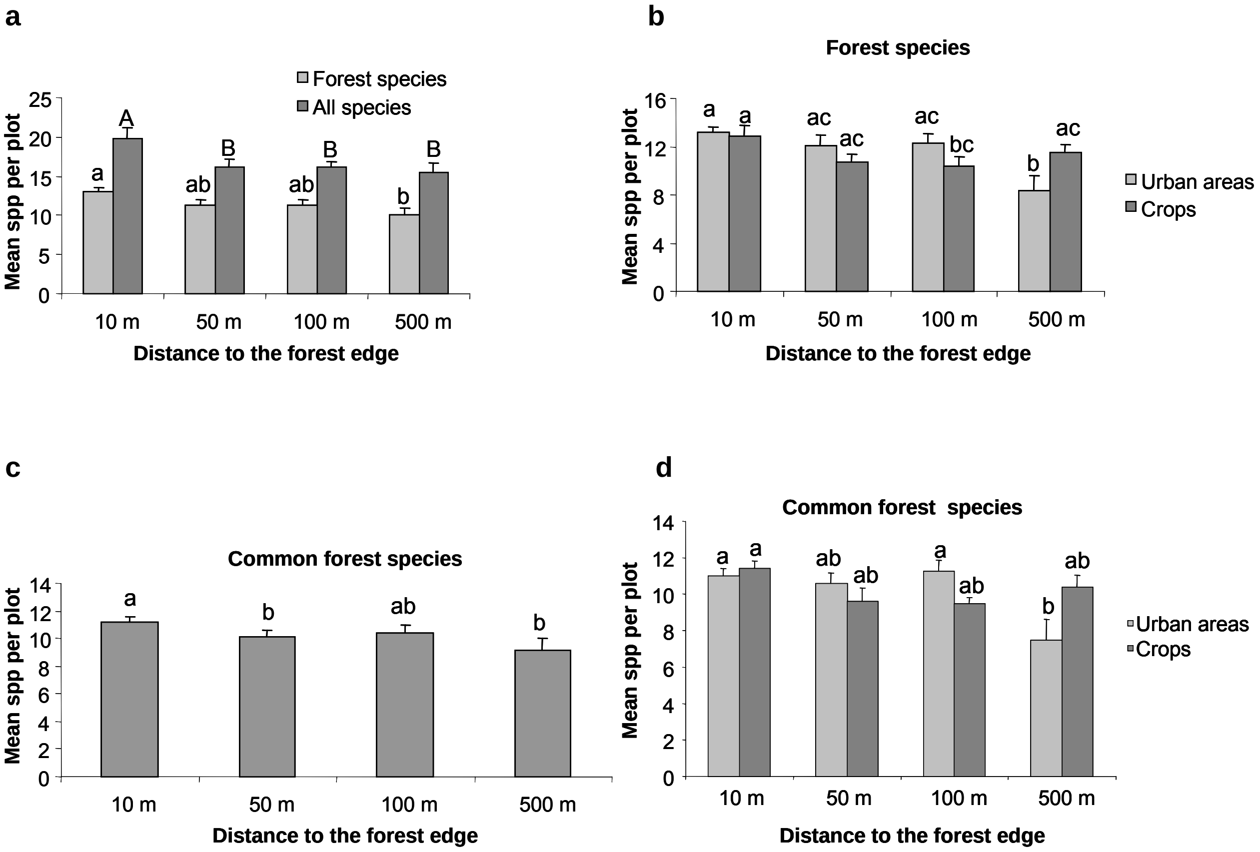

Figure 6. Large forest areas: Significant factor effects on mean species richness per plot (100 m^2) of ecological and rarity groups. (a) Distance from the forest edge on richness of all species and of forest species. (b) Interaction between distance from the forest edge and adjacent land use for forest species. For common forest species, effects of distance to the forest edge are shown in (c), and the interaction between distance from the edge and adjacent land use in (d). Error bars are one standard error of the mean. Different letters show that means are significantly different (LSD test, $P < 0.05$).

Table 2. ANOVA results for plots at between 10 to 100 m from the forest edge (i.e. excluding 500-m plots): Effects of forest-size category, adjacent land use (Adjacency) and edge distance (Edgedist) on the average richness of species groups. Significant effects (*P*<0.05) are indicated by an asterisk.

	All species			Forest species			Shrubland-grassland species			Synanthropic species			Common forest species			Rare forest species		
Factor	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*
Forest size	0.0005	0.07	0.795	0.007	1.65	0.205	0.245	4.46	0.039*	0.383	8.96	0.004*	0.009	2.09	0.155	0.111	5.918	0.019*
Adjacency	0.0176	2.57	0.115	0.004	1.00	0.322	0.024	0.43	0.514	0.088	2.06	0.158	0.017	4.08	0.049*	0.010	0.505	0.481
EdgeDist	0.008	1.11	0.339	0.001	0.14	0.873	0.001	0.01	0.984	0.137	3.21	0.049*	0.002	0.36	0.699	0.012	0.635	0.534
Forest size x Adjacency	0.020	2.97	0.091	0.001	0.20	0.658	0.016	0.30	0.588	0.200	4.68	0.036*	0.002	0.36	0.549	0.084	4.443	0.040*
Forest size x EdgeDist	0.020	2.98	0.060	0.011	2.54	0.089	0.067	1.22	0.304	0.013	0.32	0.731	0.011	2.58	0.086	0.024	1.279	0.288
Adjacency x EdgeDist	0.001	0.13	0.880	0.006	1.45	0.244	0.039	0.70	0.500	<0.001	0.01	0.993	0.001	1.33	0.274	0.005	0.277	0.759
Forest size x Adjacency x EdgeDist	0.002	0.25	0.779	0.002	0.54	0.587	0.010	0.189	0.829	0.002	0.05	0.955	0.003	0.64	0.530	0.014	0.769	0.469

Table 3. ANOVA results for large forest areas (i.e. plots at between 10 to 500 m from the forest edge): Effects of adjacent land use (Adjacency) and edge distance (Edgedist) on the average richness of species groups. Significant factors (*P*<0.05) are indicated by an asterisk.

	All species			Forest species			Shrubland-grassland species			Synanthropic species			Common forest species			Rare forest species		
Factor	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*	MS	F	*P*
Adjacency	0.017	0.882	0.350	0.00002	0.004	0.951	0.019	0.492	0.488	0.117	2.75	0.107	<0.0001	<0.001	0.986	0.011	0.235	0.629
EdgeDist	0.061	3.223	0.025*	0.022	4.412	0.011*	0.009	0.243	0.866	0.060	1.42	0.257	0.044	3.495	0.018*	0.043	0.880	0.454
Adjacency x EdgeDist	0.045	2.371	0.075	0.019	3.854	0.019*	0.015	0.387	0.763	0.056	1.32	0.287	0.048	3.794	0.012*	0.032	0.662	0.577

Human frequentation in relation to studied factors

High human frequentation was less common in forests adjacent to crops, whereas it was over represented in those adjacent to urban areas (χ^2=29.0, *P*<0.001 for both forest-size categories; χ^2=23.8, *P*<0.001 for large forest areas; Table 4). Furthermore, low human frequentation increased with distance from the forest edges, but only in plots adjacent to urban areas (χ^2=14.5, *P*<0.01 for both forest-size categories, and χ^2=18.5, *P*<0.001 for large forest areas; Table 4). No relationship was found between frequentation and distance to the edge for sites adjacent to crops, neither between frequentation and forest-size categories.

Table. 4. Association between the intensity of human frequentation and (a) adjacent land uses and (b) distance to the forest edge when adjacent to urban areas, considering either both forest-size categories (i.e. for plots at between 10 to 100 m from the forest edge, thus excluding 500-m plots), or large forests areas (i.e. including plots at between 10 to 500 m from the forest edge). Results of the four resultant contingency tables are given, all of them being significant (*P*<0.01).

Human frequentation		(a) Adjacent land use			(b) Distance to the forest edge (Adjacent to urban areas)				
		Crops	Urban areas	Total	10 m	50 m	100 m	500 m	Total
Both forest-size categories									
Low frequentation	Observed	79	46	125	7	18	22	-	47
	Expected	62.5	62.5	125	15.67	15.67	15.67	-	47
High frequentation	Observed	11	44	55	23	12	8	-	43
	Expected	27.5	27.5	55	14.33	14.33	14.33		43
	Total	90	90	180	30	30	30	-	90
Large forest areas									
Low frequentation	Observed	57	36	93	3	7	11	13	34
	Expected	46.5	46.5	93	8.5	8.5	8.5	8.5	34
High frequentation	Observed	3	24	27	12	8	4	2	26
	Expected	13.5	13.5	27	6.5	6.5	6.5	6.5	26
	Total	60	60	120	15	15	15	15	60

DISCUSSION

Fragmented peri-urban forests are characterized by processes of rarefaction of the most vulnerable forest species, and by an increasing presence of non-forest plants, both native and exotic (Honnay et al. 1999c; Hobbs 1988, 2000; Godefroid & Koedam 2003b). Forest patches in the present study also are highly affected by this process of anthropisation, as 62% of all recorded plant species are synanthropic or characteristic of open-canopy forests, shrublands or grasslands. NMS analyses suggest that the main sources of variation of understory plant composition in these forests are species of relatively open habitats (Table 1). In contrast, strict forest species (both rare and common) had much less weight in the ordination of plots indicating a relatively homogeneous pool of forest species.

Forest patch size is a key factor determining plant species richness in forest archipelagos (i.e. Peterken & Game 1984; Hobbs 1988; Dzwonko & Loster 1989, 1992; Zacharias & Brandes 1990), but this species-area relationship might vary because of direct or indirect effects of human activities (Hobbs 1988, 2001; Honnay et al. 1999c; Bennet et al. 2004), especially in urban and peri-urban contexts (Moffatt et al. 2004). Indeed, our results showed differences in the species richness per plot depending on the land use adjacent to forests and on the rarity and ecology of the species involved. Direct human activity in forests facilitates seed and propagule dispersal of non-forest species and creates open, disturbed areas, which favor these species by increasing light availability and reducing competition with forest plants (Hobbs 1988; Gilliam 1995; Deconchat & Balent 2001). Human activity affects rare forest species as well, probably inducing changes in the suitability of their habitats (Hill et al. 2001; Dumortier et al. 2002).

Many of the factors affecting forest species richness and composition have been related to processes occurring at the forest edge (Wiens 1992; Forman 1995b; Murcia 1995; Cadenasso & Pickett 2001). It has been suggested that human influences percolate into the forests from their edges, thus supporting the hypotheses that small forest patches are potentially more affected (because of their higher perimeter-area ratio). Describing edge effects is a difficult task because of the high number of factors concurring and interacting among them (Murcia, 1995). The present study illustrates, in the case of the peri-urban area of Barcelona: (i) differential effects of the analyzed factors on several ecological and rarity groups, meaning that edge effects would potentially affect understory plant species composition in these forests; and (ii) the existence of complex interactions between forest-size category, adjacent land use and distance to the edge. These issues are expanded below.

Our study indicates contrasting associations between forest–size category and species richness for diverse ecological and rarity groups, thus affecting understory species composition: indeed, richness per plot of rare forest species and of shrubland-grassland species increased in large forests *versus* small patches, whereas synanthropic species exhibited the opposite pattern. Previous studies have found an association between forest size and plant composition, large forest areas usually being less altered and containing more rare forest species (Saunders et al. 1991; Luken 1997). No significant association was found between forest-size category and frequentation in the present study, although large forest patches were less frequented than small ones in a previous study involving a different set of 147 forest patches distributed across the Vallès lowlands (Guirado 2002). Human visitation could particularly favor the percolation of synanthropic species in small forest patches because they have high perimeter-area ratios and may experience higher densities of visitors per unit area than larger patches (Laurance & Yensen 1991; Hill & Curran 2001; Ross et al. 2002).

Adjacent land use and human frequentation have been shown to modify species-area relationships and species assemblages in forests (Hobbs 1988, 2001; Honnay et al. 1999c; Jaquemyn et al. 2001; Dumortier et al. 2002; Gibb & Hochuli 2002; Bennett et al. 2004). We have found that these factors show contrasting interactions with forest-size category among ecological and rarity groups of understory plant species (Figs. 4d, 5c and 6b,d). These results suggest that the higher frequentation observed near urban areas might favor the spread of both common forest and synanthropic species as other authors have indicated (Hobbs 1988; Gilliam et al. 1995; Deconchat & Balent 2001). It might also be critical in the disappearance of rare forest species as observed in other studies (Hill & Curran 2001; Dumortier et al. 2002).

Distance to the edge has been traditionally used to predict edge effects. Species richness of different plant groups responds differently to distance from edges (Murcia 1995; Pauchard & Alaback 2004; Moffatt et al. 2004; Holt et al. 2005). We have found that this response also depends on forest-size category, being significant for small patches but not for large forest areas in the case of synanthropic plants, and *vice versa* in the case of forest plants. In general, results from edge-to-interior studies indicate that the number of ruderal and exotic species is highest at edges and tends to decline towards the interior (Fraver 1994; Pauchard & Alaback 2004). However, distance to the edge was not significant for synanthropic species richness in large forest areas of the Vallès, perhaps because these large forests have, on average, few synanthropic species (Fig 4a). In the case of forest species, large peripheral zones inside the patches are often needed to detect changes in richness and composition between edge and core in highly disturbed peri-urban areas (Ranney et al. 1981; Laurence & Yensen 1991; Honnay et al. 1999c). Similar results were found for arthropods in large and small forest patches in the Sydney region (Australia, Gibb & Hochuli 2002), confirming that the effects of human impact in urban and sub-urban forests occur deeper into the forest than in more rural forest-cropland mosaics (Matlack 1993), though patterns of variation can vary among taxonomic groups (Gibb & Hochuli 2002).

Adjacent human land uses also modulate the edge effects in the Vallès forests. Forest species richness decreased with increasing distance from the forest edge in large forests adjacent to urban areas, but not in those adjacent to crops (Fig. 6b). The same pattern was observed for common forest species richness, which was significantly higher near urban areas but not near crops (Fig. 6d). Agricultural activities and urban areas have different effects on forest edges: agricultural activity might modify some forest edge properties, causing eutrophication (Dumortier et al. 2002) and enhancing the arrival and spread of many weed and crop species into forests (Grashof-Bokdam 1997; Koerner et al. 1997). In contrast, urban land uses would determine more intense soil compaction due to pedestrian traffic, and it would promote the arrival and spread of garden and ruderal species (Godefroid & Koedam 2003b, 2004a).

Our study also highlights human frequentation as an associated factor to edge effects that could play a central role on understory plant species richness and assemblage. Frequentation was significantly concentrated near the edge of forests adjacent to urban areas, and this pattern essentially coincides with those exhibited by total forest species and common forest species, whose richness decreased as edge distance increases (Fig. 6a,c,d). This association between the patterns of frequentation and richness of some species groups in relation to the distance to the edge was detected in forests adjacent to urban areas, which are those potentially more frequented, but not in those adjacent to crops. Disturbance associated with frequentation can stimulate the germination of early successional species from the soil seed bank, as suggested by Brothers & Spingarn (1992), Deconchat & Balent (2001), and Honnay et al. (2002). Soil compaction associated to forest frequentation has been proven to determine growth reduction in some strict forest species (Godefroid & Koedam 2004a). On the other hand, frequentation might favor the establishment of some forest species by creating gaps, especially in the case of forest edge species. Further research is needed to determine which factors that could facilitate species establishment are associated with frequentation.

Implications for conservation

Our results showed that biodiversity conservation policies in fragmented metropolitan forests should prioritize the preservation of large forest areas (mainly adjacent to crops), because they are less frequented (Guirado 2002) and are also better buffered against the establishment of non-forest species, which in turn favors the preservation of rare forest species. Honnay et al. (1999c) reached similar conclusions in disturbed suburban forest patches in Belgium. There has been a long debate –not yet completely resolved- concerning whether the remnants of natural habitats should be organized around a few large, versus several small, patches (e.g. Diamond 1975; Terborgh 1976; Simberloff & Abele 1976) with the goal of maximizing species richness. Many authors consider that the conservation of several small forest patches geographically distributed may favor interpatch habitat diversity, which is greater than for few large forest areas (e.g., Peterken & Game 1984; Dzwonko & Loster 1989; Honnay et al. 1999d). In general, those studies are located in rural areas, where human pressures are lower than in peri-urban ones. In peri-urban areas, small forests act like acclimatization gardens for synanthropic species, as our results show: small forest patches had 32% more synanthropic species per plot than large forest areas. Further studies are needed in order to ascertain the role of these small forests on the establishment and eventual spread into forests of synanthropic species in urban and peri-urban areas. These studies can assist in improving the management of synanthropic species in man-modified forests, and in developing guidelines for land management oriented to avoid the expansion of these species. Meanwhile, our results might be useful for the design of conservation policies for the maintenance of forest biodiversity and conservation status in peri-urban Mediterranean areas.

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Appendices

Appendix I. Abbreviations of plant species taken into account in the study (n= 96), based on the first four letters of genus and species names.

Genus	**species**	**Abbreviation**
Agrimonia	eupatoria	Agrieupa
Amaranthus	retroflexus	Amarretr
Araujia	sericifera	Arauseri
Arbutus	unedo	Arbuuned
Arctium	minus	Arctminu
Arum	italicum	Arumital
Arundo	donax	Arundona
Asparagus	acutifolius	Aspaacut
Asplenium	adiantum-nigrum	Aspladia
Bellis	perennis	Bellpere
Bidens	subalternans	Bidesuba
Brachypodium	phoenicoides	Bracphoe
Brachypodium	sylvaticum	Bracsylv
Bryonia	cretica	Bryocret
Bupleurum	rigidum	Buplrigi
Calicotome	spinosa	Calispin
Calluna	vulgaris	Callvulg
Celtis	australis	Celtaust
Chenopodium	album	Chenalbu
Cirsium	vulgare	Cirsvulg
Cistus	albidus	Cistalbi
Cistus	monspeliensis	Cistmons
Cistus	salviifolius	Cistsalv
Clematis	flammula	Clemflam
Clematis	vitalba	Clemvita
Conyza	canadensis	Conycana
Conyza	sumatrensis	Conysuma
Coriaria	myrtifolia	Corimyrt
Cornus	sanguinea	Cornsang
Corylus	avellana	Coryavel
Cotoneaster	nebrodensis	Cotonebr
Crataegus	monogyna	Cratmono
Dactylis	glomerata	Dactglom
Daphne	gnidium	Daphgnid
Daucus	carota	Dauccaro
Dorycnium	hirsutum	Doryhirs
Erica	arborea	Ericarbo
Euphorbia	amygdaloides	Euphamyg
Fragaria	vesca	Fragvesc
Fraxinus	angustifolia	Fraxangu
Galium	aparine	Galiapar
Galium	lucidum	Galiluci
Geranium	robertianum	Gerarobe
Hedera	helix	Hedeheli
Hypericum	perforatum	Hypeperf
Inula	helenioides	Inulhele
Inula	viscosa	Inulvisc
Juniperus	oxycedrus	Junioxyc

Genus	**species**	**Abbreviation**
Laurus	nobilis	Laurnobi
Lavandula	stoechas	Lavastoe
Ligustrum	lucidum	Liguluci
Ligustrum	vulgare	Liguvulg
Lithospermum	purpurocaeruleum	Lithpurp
Lonicera	etrusca	Lonietru
Lonicera	implexa	Loniimpl
Lonicera	xylosteum	Lonixylo
Marrubium	vulgare	Marrvulg
Mercurialis	annua	Mercannu
Olea	europaea	Oleaeuro
Oryzopsis	miliacea	Oryzmili
Osyris	alba	Osyralba
Oxalis	corniculata	Oxalcorn
Parietaria	officinalis	Parioffi
Phillyrea	angustifolia	Philangu
Phytolacca	americana	Phytamer
Pistacia	lentiscus	Pistlent
Plantago	lanceolata	Planlanc
Prunus	domestica	Prundome
Psoralea	bituminosa	Psorbitu
Quercus	cerrioides	Quercerr
Quercus	coccifera	Quercocc
Quercus	ilex	Querilex
Ranunculus	bulbosus	Ranubulb
Rhamnus	alaternus	Rhamalat
Rosa	sempervirens	Rosasemp
Rosmarinus	officinalis	Rosmoffi
Rubia	peregrina	Rubipere
Rubus	ulmifolius	Rubuulmi
Ruscus	aculeatus	Ruscacul
Sanguisorba	minor	Sangmino
Satureja	calamintha	Satucala
Smilax	aspera	Smilaspe
Solanum	nigrum	Solanigr
Sonchus	oleraceus	Soncoler
Sonchus	tenerrimus	Sonctene
Sorbus	domestica	Sorbdome
Stellaria	media	Stelmedi
Taraxacum	officinale	Taraoffi
Thymus	vulgaris	Thymvulg
Ulex	parviflorus	Ulexparv
Ulmus	minor	Ulmumino
Urtica	dioica	Urtidioi
Verbena	officinalis	Verboffi
Viburnum	tinus	Vibutinu
Viola	alba	Violalba
Viola	sylvestris	Violsylv

Appendix II. List of (1) common forest species; (2) rare forest species; (3) synanthropic species; and (4) shrubland and grassland species recorded in (a) a random sampling of the Vallès lowlands, when we considered 252 sampling plots in 147 forest patches (chapters 2-3); and (b) an independent sampling of the Vallès lowlands and the adjacent foothills, when we considered 90 plots in 10 forest patches and 120 plots in large forest areas (chapter 5).

	(a) Vallès lowlands	**(b) Vallès lowlands and foothills**
1. Common forest species		
Acer campestre	1	1
Arbutus unedo	14	76
Arum italicum	19	5
Asparagus acutifolius	230	152
Asplenium adiantum-nigrum	76	115
Brachypodium sylvaticum	74	130
Bupleurum fruticosum	1	-
Buxus sempervirens	1	-
Centaurea lagascana	-	10
Clematis flammula	81	62
Clematis vitalba	74	35
Coriaria myrtifolia	6	10
Cornus sanguinea	16	10
Corylus avellana	5	4
Crataegus monogyna	130	87
Daphne gnidium	16	19
Daphne laureola	1	1
Equisetum ramosissimum	-	1
Euphorbia amygdaloides	6	18
Fragaria vesca	6	8
Galium maritimum	4	7
Genista triflora	-	1
Geum urbanum	-	2
Hedera helix	215	159
Hippocrepis comosa	3	1
Juniperus communis	1	-
Lapsana communis	1	1
Limodorum abortivum	-	1
Lonicera implexa	6	44
Osyris alba	37	15
Pinus halepensis	1	19
Pinus pinea	1	1
Pistacia lentiscus	17	42
Polypodium vulgare	4	21
Populus alba	1	-
Prunus spinosa	4	1
Quercus cerrioides	140	161
Quercus faginea	-	4
Quercus humilis	4	5
Quercus ilex	117	191
Quercus suber	4	1
Rhamnus alaternus	73	80
Rosa sempervirens	40	3
Rosa sp.	-	68
Rubia peregrina	226	193
Rubus ulmifolius	186	125
Ruscus aculeatus	177	153
Satureja calamintha	43	9
Sisymbrium officinale	-	1
Smilax aspera	109	129
Stachys officinalis	1	4
Tamus communis	-	1
Ulmus minor	20	12
Viola alba	26	26
Viola willkommii	8	-

	(a) Vallès lowlands	**(b) Vallès lowlands and foothills**
2. Rare forest species*		
Cephalantera rubra	-	1
Coronilla emerus	-	1
Cotoneaster nebrodensis	9	1
Epipactis microphylla	-	1
Evonymus europaeus	1	2
Fraxinus angustifolia	8	2
Geum sylvaticum	1	2
Lathyrus cirrhosus	-	1
Ligustrum vulgare	17	16
Lithospermum purpurocaeruleum	6	2
Lonicera etrusca	9	15
Lonicera periclymenum	1	4
Lonicera xylosteum	74	34
Origanum vulgare	2	1
Phillyrea latifolia	4	4
Prunella vulgaris	-	4
Prunus mahaleb	2	4
Pteridium aquilinum	1	5
Sambucus nigra	2	1
Saponaria officinalis	-	2
Silene nutans	1	1
Sorbus domestica	6	6
Sorbus torminalis	3	16
Tilia platyphyllos	2	1
Ulmus glabra	-	2
Viburnum tinus	17	69
Viola sylvestris	52	13

*Rare species in the context of the Catalan Countries. Some of them are common in the in sub-humid Mediterranean region.

	(a) Vallès lowlands	**(b) Vallès lowlands and foothills**
3. Synanthropic species		
Abies alba	-	1
Acacia dealbata	-	1
Ailanthus altissima	2	1
Amaranthus retroflexus	13	-
Anagallis arvensis	2	2
Adryala ragusina	1	5
Araujia sericifera	9	1
Arundo donax	5	-
Aster squamatus	2	-
Atriplex patula	1	-
Aucuba japonica	1	-
Avena barbata	1	-
Bryonia cretica	37	6
Celtis australis	101	35
Cercis siliquastrum	1	-
Chenopodium album	23	1
Cirsium vulgare	16	21
Commelina sp.	4	2
Convolvulus arvensis	3	1
Conyza canadensis	18	10
Conyza sumatrensis	49	22
Coronilla valentina	1	2
Cupressus sempervirens	-	1
Cydonia oblonga	-	1
Cynodon dactylon	1	-
Cynoglossum cheirifolium	1	-
Cynoglossum sp.	8	4
Daucus carota	20	8
Diplotaxis erucoides	2	2
Diplotaxis muralis	1	-
Echinochloa crus-galli	1	-
Echium parviflorum	1	-
Echium vulgare	4	5
Erucastrum nasturtiifolium	1	-
Euonymus japonica	2	-
Ficus carica	4	-
Galium aparine	13	-
Geranium robertianum	45	7
Geranium rotundifolium	3	1
Inula viscosa	5	3
Juglans regia	4	-
Laurus nobilis	32	36
Lepidium graminifolium	1	-
Ligustrum lucidum	11	9
Lilium sp.	-	1
Linaria spuria	2	-
Lonicera japonica	1	-
Mahonia aquifolium	1	1
Malva parviflora	1	-
Malva sylvestris	1	-
Marrubium vulgare	6	1
Mercurialis annua	5	-
Mirabilis jalapa	1	1
Opuntia Picus-barbarica	-	1
Oxalis corniculata	29	7
Parietaria officinalis	5	-
Pelargonium pelatium	-	1
Petroselinum crispum	1	-
Phoenix dactylifera	1	-
Phytolacca americana	6	1
Pittosporum tobira	2	-
Poa annua	1	-
Polygonum convolvulus	-	1
Portulaca oleracea	3	-
Prunus avium	3	4
Prunus cerasifera	1	2
Prunus domestica	37	33
Pyracantha coccinea	2	3
Robinia pseudoacacia	1	5
Rumex pulcher	4	-
Ruta chalepensis	1	3
Senecio pterophorus	3	5
Senecio vulgaris	-	1
Solanum chenopodioides	3	5
Solanum lycopersicum	2	-
Solanum nigrum	14	7
Sonchus oleraceus	14	8
Stellaria media	13	4
Taraxacum officinale	48	12
Urtica dioica	5	1
Verbena officinalis	13	-
Verbena sp.	3	-
Vicia faba	1	-
Vitis vinifera	2	4
Xanthium spinosum	4	1

	(a) Vallès lowlands	(b) Vallès lowlands and foothills
4. Shrubland and grassland species		
Agrimonia eupatoria	32	9
Andryala integrifolia	3	4
Anthyllis tetraphylla	-	2
Arctium minus	23	3
Aristolochia pistolochia	3	1
Asperula cynanchica	1	-
Astragalus monspessulanum	-	4
Avenula pratensis	1	-
Bellis perennis	14	1
Bidens subalternans	9	1
Brachypodium phoenicoides	123	68
Brachypodium retusum	3	1
Bupleurum fruticescens	1	-
Bupleurum rigidum	5	2
Calicotome spinosa	20	25
Calluna vulgaris	5	3
Carex halleriana	3	-
Carlina corymbosa	-	1
Centaurea aspera	2	1
Centaurea pectinata	2	-
Cistus albidus	7	17
Cistus monspeliensis	30	43
Cistus salviifolius	46	67
Colutea arborescens	-	1
Convolvulus althaeoides	3	1
Coris monspeliensis	2	2
Dactylis glomerata	20	15
Desmazeria rigida	-	1
Dichanthium ischaemum	2	-
Dorycnium hirsutum	21	15
Dorycnium pentaphyllum	3	21
Erica arborea	39	105
Erica scoparia	1	-
Eryngium campestre	1	-
Euphorbia nicaeensis	1	-
Euphorbia segetalis	2	-
Euphorbia serrata	2	3
Foeniculum vulgare	2	2
Fumana ericoides	1	1
Galactites tomentosa	1	1
Galium lucidum	16	18
Genista scorpius	-	1
Helichrysum stoechas	2	4
Hieracium pilosella	3	6
Hyparrhenia hirta	1	-
Hypericum perforatum	11	9
Hypochoeris radicata	-	1
Inula helenioides	5	1
Inula helvetica	1	-
Juniperus oxycedrus	18	34
Koeleria phleoides	-	2
Lagurus ovatus	-	1
Lathyrus sp.	-	2
Lavandula stoechas	9	12
Melica ciliata	-	2
Melilotus indica	1	-
Olea europaea	25	15
Ononis minutissima	3	1
Ononis spinosa	-	1
Orobanche sp.	-	1
Oryzopsis miliacea	81	9
Pallenis spinosa	2	2
Petrorhagia prolifera	-	1
Phillyrea angustifolia	17	32
Picris hieracioides	4	3
Plantago lanceolata	19	4
Plantago major	1	-
Potentilla hirta	2	-
Potentilla neumanniana	1	-
Potentilla reptans	-	1
Psoralea bituminosa	9	6
Quercus coccifera	10	8
Ranunculus bulbosus	26	3
Reichardia picroides	1	-
Rosmarinus officinalis	31	48
Rumex crispus	4	-
Rumex obtusifolius	1	-
Sanguisorba minor	6	4
Sanguisorba officinalis	1	2
Sarothamnus arboreus	-	4
Scabiosa atropurpurea	2	1
Sedum album	1	-
Sedum rupestre	1	9
Silene latifolia	-	1
Sonchus tenerrimus	27	15
Spartium junceum	-	2
Sporobolus indicus	2	-
Staehelina dubia	1	6
Stipa offneri	1	-
Teucrium chamaedrys	-	3
Teucrium polium	1	-
Thymus vulgaris	11	9
Torilis nodosa	-	1
Trifolium pratense	1	4
Trifolium repens	1	-
Ulex parviflorus	50	65
Urospermum dalechampii	-	3
Verbascum sinuatum	1	-
Vicia disperma	-	2
Vincetoxicum nigrum	1	3

Conclusions generals

Les conclusions generals del treball es divideixen en diversos apartats: el primer, es centra exclusivament en les clapes de bosc de la plana del Vallès i per tant, fa referència als capítols 1, 2, 3 i 4. El segon apartat contrasta els resultats entre les clapes de bosc de la plana del Vallès i les grans àrees forestals adjacents en contacte amb el samontà de la serralada prelitoral, i es basa en els resultats obtinguts en el cinquè capítol. Posteriorment es sintetitzen de forma esquemàtica les conclusions del conjunt del treball. Finalment, es fa un seguit de propostes de gestió i conservació dels paisatges forestals peri-urbans i s'esmenten possibles línies de recerca futures en aquest àmbit.

LES CLAPES DE BOSC DE LA PLANA

S'han seleccionat a l'atzar un total de 252 parcel·les en 147 clapes de bosc de mida variable (0.25-218 ha). En la selecció de les parcel·les, s'han exclòs expressament els 10 primers metres des del marge del bosc per tal de limitar l'efecte vora eliminant de l'estudi totes aquelles formacions vegetals del marge del bosc. Per a cada parcel·la, s'ha determinat la flora present en el sotabosc i el recobriment arbori de *Quercus* spp. i de *Pinus* spp. També s'ha valorat de forma qualitativa el grau de pertorbació humana (nivell de freqüentació i el temps transcorregut des de la darrera gestió forestal). Finalment, per a cada parcel·la, s'han determinat diverses variables ambientals, històriques i d'estructura de clapa i de paisatge a partir de dades ja existents, o bé creant-les de nou mitjançant tècniques SIG. Mitjançant aquestes dades, s'ha investigat la relació existent entre les variables ambientals, estructurals i històriques i de pertorbació sobre: (1) el recobriment arbori de *Quercus* spp. i de *Pinus* spp. per parcel·la; (b) la riquesa d'espècies total i de diversos grups ecològics i de raresa per parcel·la; (c) la resposta de les espècies a nivell individual i la composició específica de les clapes de bosc estudiades. Les conclusions obtingudes es detallen a continuació:

Recobriment de *Pinus* spp. i *Quercus* spp.

- La dinàmica històrica de les clapes de bosc és molt important a l'hora d'explicar el recobriment arbori de *Quercus* spp. i de *Pinus* spp. d'aquestes: les clapes noves (posteriors a 1956) presenten un recobriment de *Pinus* superior i de *Quercus* inferior que les clapes recentment fragmentades de grans àrees forestals. A més, les clapes ja existents l'any 1956 presenten un recobriment de *Pinus* spp. superior que les clapes recentment fragmentades.
- L'estructura espacial de les clapes de bosc també està correlacionada amb el recobriment de les espècies arbòries estudiades: les clapes més connectades entre elles presenten major recobriment de *Quercus*, mentre que els boscos més aïllats presenten major recobriment de *Pinus*. És important destacar que, en l'àrea d'estudi, la connectivitat de les clapes de bosc és independent de la seva dinàmica històrica, la qual cosa suggereix que les clapes de bosc recentment fragmentades poden patir processos de regressió si queden molt aïllades, mentre que les noves clapes de bosc poden avançar amb certa rapidesa cap a estadis successionals més avançats si estan molt connectades.
- Els factors antròpics estudiats tenen un paper molt escàs a l'hora de determinar el recobriment de *Quercus* d'aquests boscos, contràriament a l'observat per a la riquesa de plantes vasculars del sotabosc. Probablement per aquest motiu, processos que actuen a escala temporal relativament llarga (com la dinàmica històrica de les clapes de bosc i la connectivitat) tenen un paper molt més rellevant en el recobriment arbori.

Composició florística i resposta de les espècies a variables físiques, estructurals i de pertorbació.

- La composició florística de les clapes de bosc de la plana del Vallès és força heterogènia entre clapes. A més, hi abunden les espècies sinantròpiques (ruderals nadiues, exòtiques, cultivades, etc.), mentre que hi són escasses les espècies forestals rares a escala regional. La majoria d'espècies forestals són comunes.

- Es distingeixen tres grans grups d'espècies en funció de la seva resposta a les variables estudiades: espècies herbàcies i arbustives lligades a boscos ben preservats, amb un elevat recobriment arbori de Q*uercus* i sovint fragmentats de grans àrees forestals; espècies llenyoses lligades a boscos ben preservats i ben connectats; i espècies exòtiques i ruderals lligades a boscos nous o bé boscos pertorbats amb recobriments de *Pinus* elevats.
- La majoria de les plantes estudiades responen a processos que actuen a escales temporals i espacials petites. L'activitat humana directa afavoreix l'arribada i establiment d'espècies ruderals i exòtiques (sinantròpiques). Un gran nombres d'aquestes espècies són indicadores de llocs altament freqüentats o recentment gestionats (e.g., *Taraxacum officinale*, *Conyza sumatrensis*), i moltes desapareixen quan el bosc es recupera d'aquesta pertorbació. El recobriment arbori de *Quercus* spp. també és molt important a l'hora de determinar la presència d'espècies forestals (e.g., *Hedera helix, Ruscus aculeatus*). A la vegada, moltes espècies sinantròpiques indiquen recobriment arboris de *Quercus* spp. baixos (e.g., *Taraxacum officinale, Sonchus tenerrimus, Daucus carota).*
- El nombre d'espècies que responen a variables de clapa i de paisatge és molt menor, ja que són factors més indirectes que poden induir respostes temporals més llargues. No obstant, l'accessibilitat a les clapes de bosc és una variable que actua a nivell de paisatge i que afecta positivament la resposta de força espècies ruderals ,probablement perquè són boscos molt freqüentats per l'home. A la vegada, diverses espècies de bosc i màquia es veuen afavorides per una elevada connectivitat (e.g., *Erica arborea, Asplenium adiantum-nigrum*), potser degut a la seva baixa capacitat dispersiva.

Riquesa florística del sotabosc

- S’han trobat un total de 241 espècies de plantes vasculars diferents en les 252 parcel·les estudiades a la plana del Vallès. Del total d’espècies, 77 són sinantròpiques, 78 són de espècies de brolla o prat, i 67 són espècies estrictament forestals (20 de les quals considerades rares dins dels Països Catalans). La resta d’espècies (19) no s’ha pogut classificar en cap dels grups anteriorment esmentats.
- Les variables mesurades a nivell de parcel·la són les més correlacionades amb la riquesa dels diversos grups estudiats. Aquests resultats coincideixen amb els obtinguts per a moltes espècies individuals. Entre aquestes variables, el recobriment arbori de *Quercus* spp., les pertorbacions humanes directes i l’accessibilitat potencial de l’home dins del bosc tenen un paper molt destacat, principalment pel què fa a la riquesa d’espècies no forestals.
- Els boscos amb un elevat recobriment arbori de *Quercus* spp. tenen menys espècies sinantròpiques i més espècies forestals que aquells amb un recobriment molt menor. Aquest fet està relacionat amb una menor entrada de llum en alzinars ben desenvolupats, així com una menor freqüentació humana dins d’aquests boscos generalment poc transitables.
- Els boscos recentment gestionats, o bé amb una elevada freqüentació humana, presenten una riquesa d’espècies sinantròpiques elevada. En canvi, la riquesa d’espècies forestals està poc determinada per qualsevol de les variables estudiades. Aquests resultats manifesten una vegada més que, en una àrea peri-urbana com la plana del Vallès, l’activitat de l’home afavoreix la presència d’espècies no forestals (especialment sinantròpiques) creant les condicions adequades per a elles. A la vegada, les espècies forestals que persisteixen als fragments de la plana són relativament resistents als processos de pertorbació i fragmentació, probablement perquè les espècies forestals més sensibles van desaparèixer ja fa temps.
- L’estructura del paisatge (e.g., distància al marge, ús del sòl adjacent) també està relacionada amb la riquesa d’algun grups estudiats, ja que pot modular l’accessibilitat

potencial de l'home dins del bosc: les parcel·les més accessibles estan, en general, més pertorbades i, per tant, tenen una major riquesa d'espècies sinantròpiques. A la vegada, les parcel·les més properes a àrees urbanes són més accessibles, la qual cosa afavoreix la penetració d'espècies sinantròpiques a través dels marges del bosc. També per aquest motiu les parcel·les més properes al marge del bosc tenen una riquesa d'espècies d'aquest grup superior. L'àrea de les clapes de bosc juga un paper poc important en la riquesa d'espècies per parcel·la.

La pertorbació antròpica i la gestió

- Els nivells de freqüentació humana a la plana del Vallès vénen explicats per diversos factors territorials que actuen a diferents escales espacials: a escala regional s'observa que la freqüentació humana és més alta a l'oest de la zona d'estudi, coincidint amb una major influència de la conurbació de Barcelona. A escala local, les parcel·les més properes al marge del bosc també estan més freqüentades que les parcel·les més interiors degut a la seva major accessibilitat.
- Aquesta relació entre la distància al marge del bosc i el nivell de freqüentació humana determina, a la vegada, efectes sobre la composició específica d'aquests boscos, tal i com s'ha observat en els apartats precedents. De forma general s'espera que els marges de les clapes de bosc de la plana del Vallès, així com aquelles clapes més properes a la conurbació barcelonina, estiguin més exposades a processos d'extinció d'espècies forestals rares i a la invasió d'espècies exòtiques i ruderals.
- L'àrea dels fragments de bosc de la plana del Vallès està associada de forma directa als nivells de gestió forestal i de freqüentació humana, la qual cosa suggereix, de nou, que la mida del bosc pot passar a segon terme en paisatges altament fragmentats i antropitzats. A més, cap de les variables estudiades determina el grau de gestió forestal, probablement perquè no depèn tant de l'accessibilitat o de la qualitat del bosc, sinó de l'ús que cada propietari en fa d'ell.

LA PLANA DEL VALLÈS I EL SAMONTÀ ADJACENT: CLAPES DE BOSC PETITES *VERSUS* GRANS ÀREES FORESTALS

S’han distribuït 20 punts de mostreig en dues categories de mida de bosc: clapes de bosc petites (8-90 ha), localitzades a la plana del vallès i grans àrees forestals (> 18,000 ha) localitzades a la plana del vallès i al samontà adjacent a la plana. Per a cada categoria de mida de bosc, 5 punts de mostreig s’han localitzat al costat de camps de cultiu, i 5 adjacents a àrees urbanes. Posteriorment, s’han determinat les plantes vasculars presents al sotabosc i el grau de freqüentació humana en 210 parcel·les de 100 m^2 localitzades a 10, 50 i 100 m del marge del bosc i, de forma addicional, a 500 m en el cas de les grans àrees forestals. En la selecció de les parcel·les s’han exclòs expressament els 10 primers metres des del marge del bosc per evitar l’efecte de vora més immediat. Mitjançant aquestes dades, s’ha investigat la composició florística dels boscos estudiats. També s’ha estudiat la relació existent entre les categories de mida de bosc, la distància al marge, l’ús del sòl adjacent (camp de cultiu o àrea urbana) i la interacció entre aquestes variables a l’hora de determinar la riquesa d’espècies total i de diversos grups ecològics i de raresa per parcel·la. Les conclusions obtingudes es detallen a continuació:

Riquesa florística del sotabosc

- S’han trobat un total de 203 espècies de plantes vasculars diferents en les 210 parcel·les estudiades als boscos de la plana del Vallès i al samontà adjacent. Del total d’espècies, 51 són sinantròpiques, 75 són de espècies de brolla o prat, i 77 són espècies estrictament forestals (27 de les quals considerades rares dins dels Països Catalans).
- Quan es consideren conjuntament les clapes de la plana del Vallès i les grans àrees forestals adjacents, la composició florística del bosc presenta pautes de variació similars a les observades per als fragments de la plana: la principal font de variació

correspon a les espècies d'hàbitats oberts (ruderals i exòtiques sobretot), mentre que les estrictament forestals varien molt menys entre parcel·les.

- L'àrea de bosc esdevé un factor clau a l'hora de determinar la riquesa de plantes vasculars del sotabosc quan es consideren les clapes de la plana i les grans àrees forestals, contràriament als resultats obtinguts quan només s'han considerat els fragments de la plana. La riquesa d'espècies de brolla i de prat i la d'espècies forestals rares és superior en grans àrees forestals que en clapes de bosc petites, mentre que la d'espècies sinantròpiques segueix un patró invers.
- De nou s'observa que, en un context fortament antropitzat com el del Vallès, l'ús del sòl adjacent i la distància al marge també són factors importants a l'hora de determinar la riquesa d'alguns grups de raresa i ecologia: les clapes de bosc petites adjacents a camps de cultiu tenen una riquesa d'espècies sinantròpiques superior que en el cas de les grans àrees forestals, però deixa de ser significatiu en boscos adjacents a àrees urbanes. L'elevada freqüentació humana observada en boscos propers a àrees urbanes (i especialment en clapes de bosc petites), afavoreix l'entrada i dispersió d'espècies sinantròpiques i, alhora, afecta negativament la persistència d'espècies forestals rares.
- L'efecte marge també interacciona amb la mida del bosc: la riquesa d'espècies sinantròpiques disminueix en augmentar la distància al marge només en clapes de bosc petites (en els primers 100 m), mentre que la riquesa d'espècies forestals disminueix en augmentar la distància al marge només en grans àrees forestals. L'efecte pertorbador antròpic s'endinsa molt més cap a l'interior del bosc quan aquest és adjacent a zones urbanes que en entorns menys pertorbats, fent que els boscos petits tinguin, en general, una composició d'espècies forestals molt homogènia. Aquests resultats suggereixen que en àrees tant humanitzades com el Vallès són necessàries distàncies més grans del marge al centre del bosc per a detectar diferències en la riquesa d'espècies forestals. Aquestes distàncies es troben, principalment, a les grans àrees de bosc.

SÍNTESI DELS RESULTATS

L'anàlisi de la composició florística dels boscos de la plana del Vallès i del samontà adjacent (Apèndix II) suggereix que aquesta és una zona és una transició entre les formacions forestals mediterrànies típiques i les submediterrànies i eurosiberianes. Destaquem la coexistència d'espècies pròpies dels boscos esclerofil·les mediterranis (p.e. *Asparagus acutifolius, Quercus ilex, Ruscus aculeatus, Rubia peregrina, Smilax aspera,* etc.) amb d'altres amb un evident component extramediterrani (*Crataegus monogyna, Lonicera xylosteum, o fins i tot Hedera helix* i *Q. cerrioides*). Sobta la baixa ocurrència d'espècies teòricament característiques del sotabosc de l'alzinar (*Viburnum tinus*). També és destacable la poca freqüència de plançons de pi (*Pinus halepensis, P. pinea*).

La Taula 1 fa una síntesi de les variables estudiades que operen als boscos vallesans i les seves conseqüències sobre els nivells de freqüentació humana, la riquesa d'espècies de diversos grups estudiats, i els recobriments arboris de *Quercus* i de *Pinus*. De l'anàlisi d'aquesta taula es desprèn que l'estat de la vegetació dels boscos vallesans està relacionat amb un conjunt de variables físiques i biològiques, però també d'estructura del paisatge i de pertorbació antròpica. Aquestes variables operen, a més, a escales espacials (paisatge, clapa, i lloc o parcel·la han estat les analitzades aquí) i temporals molt diverses, i no afecten per igual els atributs de composició i estat de a vegetació estudiats en aquest treball.

Cal destacar el fet que la mida dels fragments de bosc no ha resultat ser una variable molt explicativa pel que fa a la riquesa d'espècies per parcel·la dels diferents grups estudiats a les clapes de bosc de la plana del vallès, mentre que ha estat força important quan s'han comparat les clapes de bosc de la plana amb les grans àrees forestals del samontà adjacent. Aquests resultats suggereixen que les clapes de bosc la plana del Vallès són relativament petites per a què hi hagi diferències en la riquesa d'espècies presents per parcel·la en funció del rang d'àrees existent, i que les diferències en la riquesa venen principalment determinades per variables més qualitatives. Les pertorbacions antròpiques directes prenen un paper molt important en aquest context peri-urbà, i són probablement responsables de la substitució de les espècies forestals més exigents per plantes d'ecologia més ruderal.

PROPOSTES DE GESTIÓ I DE RECERCA FUTURES

Dels resultats obtinguts es desprenen un seguit de propostes encarades a millorar la gestió d'aquestes àrees tant fragmentades i pertorbades per l'activitat humana:

- Afavorir la preservació de grans àrees forestals allunyades el màxim possible de zones urbanes, ja que estan menys freqüentades per l'home i més protegides de l'entrada d'espècies associades a l'activitat humana. A més, la presència de grans àrees forestals també afavoreix la preservació d'espècies forestals rares.
- Mantenir les àrees urbanes el més allunyades possibles del marge del bosc, per tal de limitar l'arribada i establiment d'espècies sinantròpiques.
- Preservar les connexions de les clapes de bosc de la plana amb els grans boscos adjacents. Aquest fet permetrà: (1) aconseguir taques grans, menys susceptibles a la fragmentació; i (2) que les espècies forestals (algunes d'elles rares) presents a les grans àrees de bosc adjacents puguin arribar a colonitzar les clapes de bosc de la plana.

No obstant, manquen molts estudis que permetin determinar amb més exactitud quina és la situació en aquests boscos i quines dinàmiques actuen sobre la seva flora. Algunes propostes futures podrien ser (i) Estudiar la importància del recobriment dels diferents grups d'espècies estudiats; (ii) dur a terme estudis poblacionals i metapoblacionals sobre espècies seleccionades; (iii) analitzar com afecta la fragmentació al manteniment de les espècies forestals rares i (iv) incloure noves àrees d'estudi, en contextos similars o no, per a fer un estudi comparatiu a escala de Catalunya dels efectes de la fragmentació en la pertorbació antròpica sobre els ecosistemes forestals.

Taula 1. Principals efectes significatius que es desenvolupen a: (a) les clapes de bosc de la plana del Vallès; (b) les clapes de bosc *versus* les grans àrees forestals del samontà adjacent a la plana i que actuen a tres escales espacials diferents (parcel·la, clapa i paisatge). Les conseqüències d'aquests efectes es valoren com a atributs de qualitat del bosc, i estan integrats per: (1) el nivell de freqüentació humana; (2) la riquesa d'espècies sinantròpiques; (3) la riquesa d'espècies forestals; 4) el recobriment de *Pinus* spp. i de *Quercus* spp.; i (5) la riquesa d'espècies forestals rares. * indica la interacció entre factors; (+) relació positiva o augment del valor; (-) relació negativa o disminució del valor.

	Atributs de qualitat de bosc	**Escala espacial**		
		Parcel·la (lloc)	**Clapa**	**Paisatge**
(a) Plana	(+) Freqüentació humana	(-) distància al marge		(+) Coordenades oest-est
	(+) Riquesa espècies sinantròpiques	(-) Recobriment arbori *Quercus* spp. (+) Gestió forestal recent (-) Distància marge (+) Freqüentació antròpica alta	(+) Àrea clapa	(-) Connectivitat (+) Accessibilitat
	(+) Riquesa espècies forestals	(+)Recobriment arbori *Quercus* spp. (+) Distància pistes		
	(+)Recobriment arbori *Pinus* spp.			(-) Clapes fragmentades de grans àrees forestals després de 1956
	(+) Recobriment arbori *Quercus* spp.			(+) Clapes fragmentades de grans àrees forestals després de 1956 (+) Fragments de bosc ja existents el 1956 (+) Connectivitat
(b) Plana *vs* samontà	(+) Freqüentació humana			(-) boscos adjacents a camps de cultiu (+) Boscos adjacents a àrees urbanes
	(+) Riquesa espècies sinantròpiques	(-) Distància marge	(-) Àrea clapa	(+)Boscos adjacents a àrees urbanes
	(-) Riquesa espècies sinantròpiques		(+) Àrea clapa * (+) boscos adjacents a camps de cultiu	
	(+) Riquesa espècies forestals	(+) distància al marge (considerant clapes a 500 m del marge en grans àrees forestals)		
	(+) Riquesa espècies forestals rares		(+) Àrea clapa	
	(-) Riquesa espècies forestals rares		(-) Àrea de la clapa * (+) boscos adjacents a àrees urbanes	

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