Now that the increasing impacts and costs of invasive species are being recognized, management of alien species has become an important challenge and a high priority in biodiversity conservation. In Spain, management information on alien plants is quite scattered and in general, there is not much communication between environmental managers and scientists. Many times this is because research on the ecology and management of alien species is insufficient or difficult to access by managers.

The general aim of this thesis is to investigate management actions on alien plant species and the main limitations of these measures. Specifically, we identify the most problematic alien plant species in Spain, and we assess the main criteria used to prioritize their management. This thesis also identifies and ranks potentially invasive species in Spain, not yet present in the wild.

Globally, we conduct a meta-analysis on the impacts of invasive plant species and the consequences of their removal on native species richness and abundance. As a case study, we evaluate the efficacy of the manual removal of *Carpobrotus* in coastal Andalucía, and the recovery of the native vegetation after the invasive plant removal.

CREAF icta Estación Biológica Doñana CSIC UAB Universitat Autònoma de Barcelona

Jara Andreu Ureta 2011 Management of alien plants in Spain: from prevention to restoration

Ph.D. Thesis

# Management of alien plants in Spain: from prevention to restoration

## Gestió de plantes exòtiques a Espanya: de la prevenció a la restauració

**Ph.D. Thesis: Jara Andreu Ureta**
**September 2011**

Supervisor: Montserrat Vilà Planella
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Universitat Autònoma de Barcelona

# Management of alien plants in Spain: from prevention to restoration

Ph.D. Thesis

# Gestió de plantes exòtiques a Espanya: de la prevenció a la restauració

Memòria presentada per:

**Jara Andreu Ureta**

per a optar al grau de Doctora

Amb el vist-i-plau de:

**Directora: Montserrat Vilà Planella**

**Tutor: Francisco Lloret Maya**

Programa de Doctorat en Ciència i Tecnologia Ambientals

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Als meus pares,
per tot el seu suport

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# Agraïments

Ja fa gairebé sis anys que vaig arribar al CREAF i tot i que en un principi la tesi doctoral no era el meu objectiu, estic molt contenta d'haver aconseguit acabar aquest projecte. Treballar al CREAF ha estat i segueix sent un plaer perquè des del principi m'hi he sentit com a casa i sempre que l'he necessitat, he rebut ajuda d'un munt de persones d'aquesta casa.

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# Capítol 1

# Introducció general

## Les invasions biològiques

L'home ha transportat espècies fora de la seva àrea de distribució original, ja sigui intencionada o involuntàriament, des de temps immemorials. No obstant això, en les últimes dècades factors com la globalització ecològica i econòmica han contribuït a accelerar vertiginosament el ritme d'introducció i d'establiment d'espècies exòtiques (McNeely et al. 2001, Kowarik 2003, Levine & D'Antonio 2003). Algunes d'aquestes espècies no només aconsegueixen establir-se sinó que, a més, inicien un procés de colonització del medi, expandint-se a través de àrees extenses en un període de temps curt (Pyšek et al. 2004). Aquestes espècies són anomenades **espècies invasores** i en aquest procés d'expansió, algunes d'elles, poden ocasionar impactes considerables, tant ecològics com socioeconòmics.

En conseqüència, les invasions biològiques constitueixen, en l'actualitat, un component important del canvi global i una amenaça seriosa per a la conservació de la biodiversitat i dels ecosistemes naturals (Vitousek et al. 1997, Mack et al. 2000, Sala et al. 2000, Brooks et al. 2004, Thuiller et al. 2007). Diversos estudis asseguren que la magnitud d'aquesta amenaça continuarà creixent mentre el comerç internacional i el turisme segueixin augmentant i, mentre el clima i els usos del sòl continuïn canviant (Perrings et al. 2005, Hulme 2009, Walther et al. 2009, Vilà & Ibáñez 2011).

Entre els impactes ecològics causats per les espècies invasores podem destacar la homogeneïtzació de la biodiversitat global, a través de la disminució de la diversitat nativa local per competència, herbivoria, depredació, transmissió de malalties, hibridació, etc. (Mack et al. 2000, Sala et al. 2000, Blackburn et al. 2004, Gaertner et al. 2009, Kettunen et al. 2009). A més, les espècies invasores també poden produir canvis en l'estructura, el funcionament i els serveis dels ecosistemes (Vitousek 1994, Charles & Dukes 2008, Hejda et al. 2009, Kettunen et al. 2009, Pejchar & Mooney 2009, Erhenfeld 2010, Vilà et al. 2010), a través de, per exemple, canvis en els règims de pertorbacions (D'Antonio & Vitousek 1992, D'Antonio et al. 1999, Brooks et al. 2004, Pauchard et al. 2008), el cicle de l'aigua (Williams & Baruch 2000, Gerlach 2004, Potts et al. 2008) o els cicles biogeoquímics (Liao et al. 2008, Raizada et al. 2008, van Kleunen et al. 2010, Vilà et al. 2011).

Els impactes causats per les espècies invasores, però, no queden restringits al medi natural, sinó que també tenen repercussions en l'economia, la societat i la salut pública (Soulé 1992, Born et al. 2005, Pimentel et al. 2005, Lovell et al. 2006, Olson 2006, Kettunen et al. 2009). Els costos econòmics ocasionats per aquestes espècies poden arribar a ser enormes, ja sigui per pèrdues directes en diferents sectors econòmics com per exemple l'agricultura, la pesca o la navegació, o pels costos indirectes de la seva gestió (Pimentel et al. 2005, Kettunen et al. 2009). Als EUA s'ha estimat que les pèrdues directes ocasionades per espècies invasores i les malalties emergents conjuntament amb els costos del seu control arriben als 120 bilions de dòlars anuals (Pimentel et al. 2005). A Europa, segons un informe de la Comissió Europea (Kettunen et al. 2009), s'ha estimat que aquesta xifra assoleix com a mínim els 12,5 bilions d'Euros anuals. A Espanya, el cas de la invasió del jacint d'aigua (*Eicchornia crassipes*) al riu Guadiana mereix una atenció especial per la enorme quantitat de diners invertida en la seva retirada (~7.000.000 € anuals) i pels danys ocasionats en diferents sectors com la pesca o la navegació.

En concret, les espècies invasores poden, entre d'altres, disminuir la producció de les collites i de fusta, reduir la qualitat de les pastures, afectar vies de comunicació, obstruir infraestructures de canalització o irrigació, reduir la disponibilitat d'aigua, o bé, disminuir el valor estètic i recreatiu de les àrees naturals envaïdes. Pel que fa als impactes sobre la salut humana cal destacar que algunes malalties com per exemple al·lèrgies o dermatitis són causades per espècies exòtiques (ex. *Heracleum mantegazzianum*, *Ambrosia artemisiifolia* o *Cortaderia selloana*; Kettunen et al. 2009) i que, a més, aquestes espècies poden ser portadores de paràsits o patògens d'humans o d'animals domèstics, augmentant així la transmissió d'algunes malalties infecciones (Pimentel et al. 2005).

Totes aquestes repercussions estan adquirint gran rellevància internacional i cada vegada són més els països que adopten mesures de gestió per minimitzar les conseqüències negatives de les espècies exòtiques invasores.

## *El procés d'invasió: definicions*

Per poder gestionar les invasions biològiques és important conèixer el procés pel qual una espècie exòtica esdevé invasora. Aquest, és el resultat d'una seqüència d'estadis successius (Kolar & Lodge 2001, Leung et al. 2002) que ha estat il·lustrat a la Figura 1.1 pel cas concret de les plantes, principal objecte d'estudi d'aquesta tesi. Cal tenir en compte però, que les diferents etapes d'aquest procés no tenen perquè ocórrer en tots els casos, ja que només una petita proporció de les espècies aconsegueixen passar als estadis següents (Kolar & Lodge 2001). Per tant, del total d'espècies que són transportades a una nova regió només una petita proporció aconsegueixen esdevenir invasores.

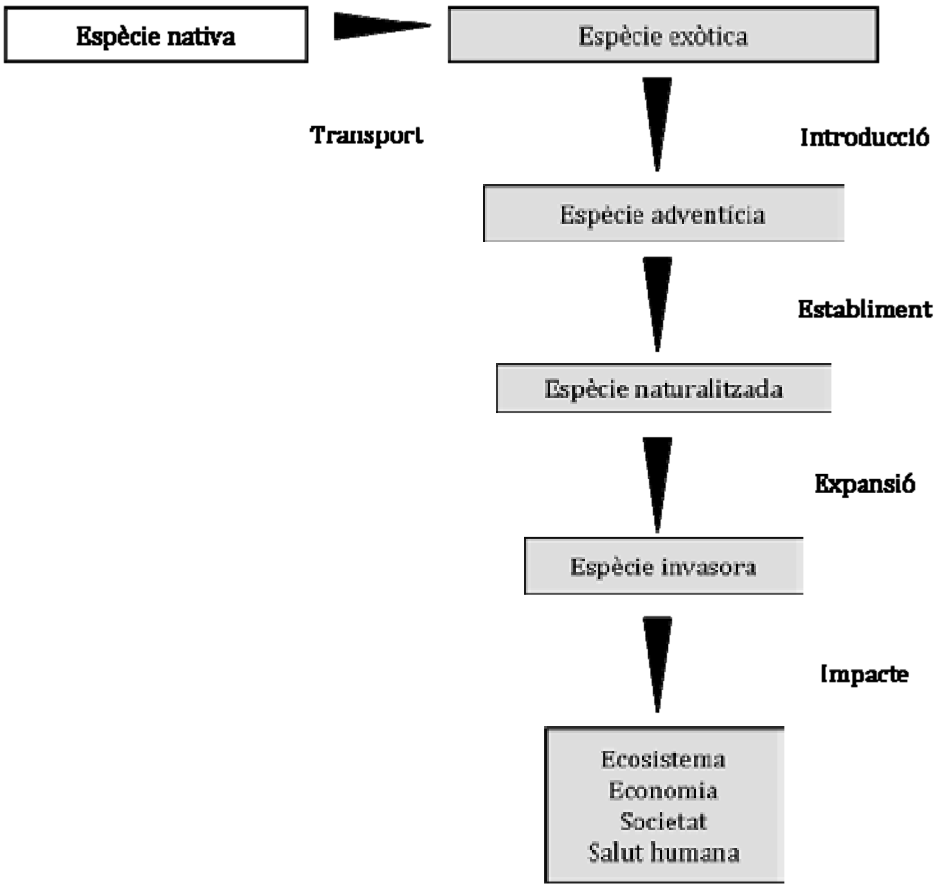



**Figura 1.1** Esquema del procés d'invasió adaptat de Pyšek et al. (2004) i Duncan et al. (2003).

A continuació s'explica cada un dels estadis del procés d'invasió de les plantes exòtiques i s'especifiquen les diferents definicions que han rebut els grups d'espècies que integren els diversos passos del procés d'invasió:

**Transport:** L'espècie ha de ser transportada des de la seva àrea nativa fins a una nova localitat. En aquesta nova localitat l'espècie passa a anomenar-se **espècie exòtica** (*Sinònims*: al·lòctona, introduïda, no nativa, forana), que no és res més que una espècie present en una determinada regió (ex. continent, illa, ecoregió o qualsevol unitat administrativa) degut a la intervenció de l'home (Pyšek et al. 2004), o que hi ha arribat sense l'ajuda humana a partir d'una àrea on no és nativa (Hulme et al. 2008b). Les vies de transport d'aquestes espècies són molt diverses, però es poden dividir, d'una manera senzilla, en intencionades i involuntàries:

- **Intencionades:** l'espècie és introduïda amb una finalitat determinada de manera legal o clandestina. Per exemple, les plantes poden ser introduïdes pel seu ús agrícola, forestal, ornamental, per a la protecció de sòls, per l'obtenció de medicaments, fibres o matèries primeres per la indústria, etc.

- **Involuntàries:** l'espècie és introduïda de forma no intencionada per l'home a través, per exemple, del transport de mercaderies (ex. a les rodes dels camions), de llavors contaminants dels productes agrícoles, de moviments de terres, etc. (Hulme 2005, Hulme et al. 2008b).

**Introducció:** L'espècie ha de ser introduïda en el nou ambient. En el cas que provingui d'un transport involuntari, la distinció entre aquests dos primers estadis del procés és pràcticament impossible, però en el cas que provingui d'un transport intencionat, la via d'introducció al medi natural es pot donar mitjançant un alliberament deliberat (ex. abocament de restes de jardineria) o accidental (ex. dispersió de llavors des dels jardins; Hulme et al. 2008b). Una vegada introduïdes al medi natural les espècies exòtiques s'anomenen **espècies adventícies o no establertes** (*Sinònims*: subespontànies, "casuals" en anglès). Són espècies exòtiques que poden reproduir-se ocasionalment fora de l'àrea de conreu, però que acabarien desapareixent perquè no són capaces de formar poblacions autosostenibles, ja que necessiten reintroduccions constants per tal de sobreviure (Pyšek et al. 2004).

**Establiment:** Després de la introducció, l'espècie pot aconseguir establir-se i passa a ser anomenada **espècie naturalitzada o establerta**, és a dir, capaç de mantenir poblacions autosostenibles, com a mínim durant 10 anys, sense la intervenció humana directa (Pyšek et al. 2004).

**Expansió:** Les espècies que aconsegueixen establir-se amb èxit poden augmentar en abundància i propagar-se més enllà del punt d'introducció. Aquestes espècies poden considerar-se invasores. A la literatura hi ha un ús desigual del terme **espècie invasora**. Principalment, existeixen dos punts de vista: un basat en la capacitat d'expansió i un altre en la capacitat d'ocasionar un impacte. La primera definició, majoritàriament utilitzada pels ecòlegs, considera que les espècies invasores són espècies naturalitzades en ambients naturals o seminaturals que produeixen descendència reproductiva, sovint en grans quantitats, a distàncies considerables dels individus parentals, i que tenen el potencial per expandir-se en grans àrees (Pyšek et al. 2004). L'altra definició és més restrictiva, ja que considera que les espècies invasores són aquelles espècies naturalitzades que produeixen canvis significatius en la composició, estructura o processos dels ecosistemes. Aquesta definició s'utilitza en organismes nacionals com el GAE (*Grupo de Aves Exóticas*) o el GEIB (*Grupo Especialista en Invasiones Biologicas*), i en institucions internacionals com per exemple la Unió Internacional per la Conservació de la Natura (IUCN; http://www.issg.org/) o a la Convenció de la Biodiversitat (CBD; http://www.cbd.int/). En aquest treball adoptarem la primera definició basada en el criteri geogràfic ja que creiem que considerar el impacte ambiental per decidir si una espècie és invasora o no, resulta poc operatiu. Molt sovint, el impacte real es desconeix perquè no s'ha quantificat o perquè és difícil d'estimar (Vilà et al. 2010).

### *Factors que determinen l'èxit d'invasió*

Predir l'èxit i el impacte de les espècies exòtiques consisteix en determinar el **risc d'invasió** i ha estat un dels motors de la recerca ecològica sobre invasions biològiques (Elton 1958, Reichard & Hamilton 1997, Lonsdale 1999). L'èxit d'invasió de les espècies exòtiques depèn de l'èxit invasor de l'espècie en concret i del grau d'invasió, és a dir, de la seva abundància al territori d'introducció. S'han suggerit diversos factors no excloents que podrien estar relacionats amb l'èxit d'invasió (Figura 1.2).

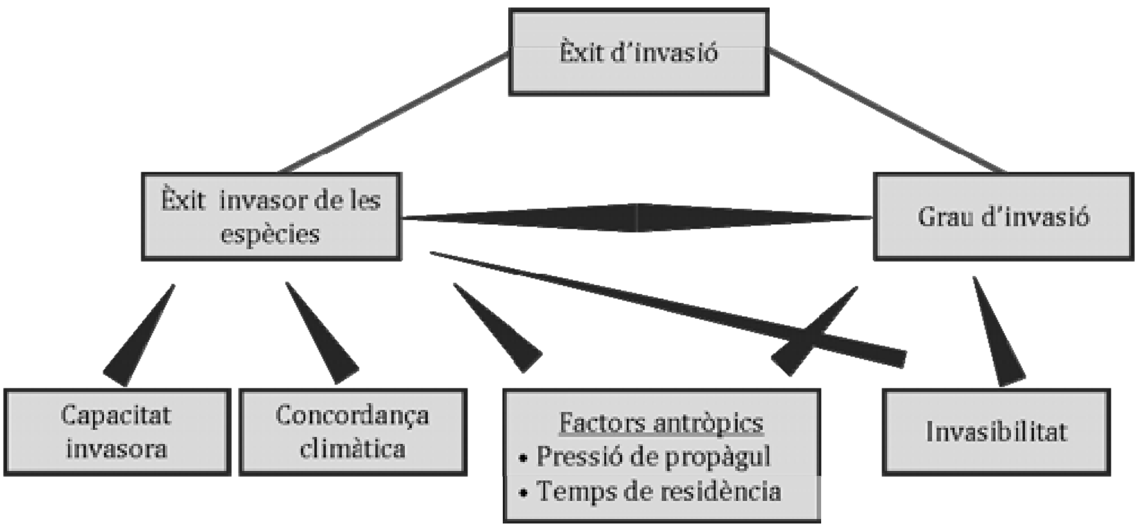



**Figura 1.2** Esquema dels components de l'èxit d'invasió i els seus factors d'influència. Modificat a partir de Gassó (2008).

Per una banda, l'èxit invasor de les espècies invasores està influenciat principalment per la seva capacitat invasora i per la concordança climàtica (Figura 1.2). La **capacitat invasora** o **potencial invasor** es defineix com la capacitat intrínseca d'una espècie per colonitzar, expandir-se i esdevenir invasora (di Castri 1989). Molts estudis han tractat de conèixer quins trets biològics determinen que certes espècies manifestin més tendència a l'èxit. No obstant, s'ha demostrat que identificar aquests trets és força difícil (Pyšek & Richardson 2007). Alguns dels trets biològics i ecològics de les espècies invasores identificats fins ara són: capacitat competitiva elevada, creixement ràpid, fecunditat elevada, dispersió eficaç, genoma reduït, o superfície foliar específica elevada, entre d'altres (Rejmánek 1996, Rejmánek & Richardson 1996, Lake & Leishman 2003).

L'altre factor d'influència sobre l'èxit invasor és la **concordança climàtica**, que fa referència a que les espècies haurien de tenir millors oportunitats d'establir-se si el clima i les condicions físiques del lloc d'introducció i de l'àrea de distribució nativa de l'espècie coincideixen (Brown 1989, Panetta & Mitchell 1991, Scott & Panetta 1993, Williamson 1996). Per tant, les espècies de latituds o regions biogeogràfiques similars tindran més èxit (Sol et al. 2005). Hi ha diversos estudis que reforcen aquesta hipòtesi (Curnutt 2000, Blackburn & Duncan 2001b, Cassey 2003, Hayes & Barry 2008), i que fins i tot la utilitzen com a punt de partida per a models de risc d'invasió (Thuiller et al. 2005).

Per altre banda, les variacions en el grau d'invasió entre localitats podrien ser degudes, entre d'altres factors, a diferències en la **invasibilitat**, és a dir, la resistència que l'ecosistema receptor ofereix a la invasió (Figura 1.2). Per saber si una comunitat o hàbitat és més o menys envaïble ens hem de preguntar quins factors abiòtics i biòtics limiten la germinació de les llavors i l'establiment de plàntules de l'espècie en aquest lloc (Lonsdale 1999). La diversitat d'espècies natives, les interaccions interespecífiques (mutualismes, competència, herbivoria, etc.), les pertorbacions i la disponibilitat de nutrients al sòl han estat els factors més examinats per detectar diferències en la invasibilitat. S'ha proposat que l'eutrofització del sòl i la disponibilitat de sòls erms són els factors que ofereixen una major invasibilitat (Kowarik 1990, Burke & Grime 1996, Lonsdale 1999, Hobbs 2000).

Finalment, la **pressió de propàguls** i el **temps de residència** són els altres dos factors que determinen l'èxit d'invasió (Blackburn & Duncan 2001a, Duncan et al. 2003; Figura 1.2). La pressió de propàguls o esforç d'introducció (Blackburn & Duncan 2001a) és una mesura composta pel nombre d'individus d'una espècie alliberats en una regió i la freqüència en què han tingut lloc les introduccions (Carlton 1996). El temps de residència és el temps des de la introducció; com més temps fa que l'espècie s'ha introduït més gran és la mida del banc de propàguls, i més gran la probabilitat de dispersió i establiment de noves poblacions (Rejmánek et al. 2005b). El impacte ecològic d'una espècie introduïda també augmenta amb el temps de residència (Collier et al. 2002).

## Gestió de les invasions biològiques

Degut als impactes ecològics i socioeconòmics causats per les invasions biològiques, la seva gestió s'ha convertit en una prioritat pels agents mediambientals, sobretot als espais protegits. L'enfocament estratègic més adient per a la gestió d'aquestes espècies inclou tres etapes, que poden ser complementàries: 1) prevenció, 2) detecció ràpida i actuació immediata mitjançant eradicació, 3) contenció i mesures de control acompanyades de mitigació d'impactes (Rejmánek 2000, Hulme 2006, NISC 2008). El manteniment, la restauració i el seguiment del bon estat ecològic dels ecosistemes afectats, tot i que no es poden considerar mètodes de gestió en sí mateixos, són necessaris una vegada s'han dut a terme actuacions d'eradicació, contenció o control d'una determinada espècie.

L'adopció d'una o un altre estratègia de gestió dependrà, en gran mesura, de l'estadi en què es trobi la invasió biològica (Figura 1.3), així com de les possibilitats reals d'èxit d'acord amb les característiques del medi i de l'espècie, dels recursos disponibles, del suport institucional i de la distribució dels esforços de gestió al llarg del temps (Hulme 2006).

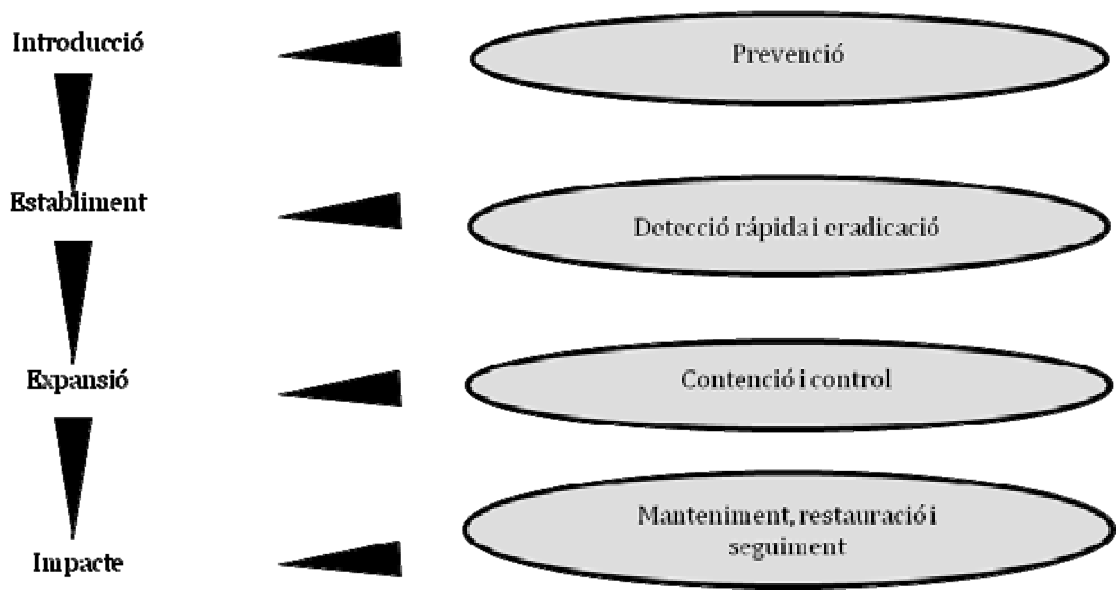



**Figura 1.3** Relació entre les diferents etapes del procés d'invasió i les possibles estratègies utilitzades per a fer front a les invasions biològiques. Esquema adaptat de Hulme (2006).

## *Prevenció*

La prevenció suposa evitar l'entrada d'espècies potencialment invasores a una nova regió. Donat que generalment el control o l'eradicació són difícils i molt costosos, la prevenció hauria de ser el recurs principal en la lluita contra les invasions biològiques (Hulme 2006). A més, és l'estratègia més eficaç, amb menor cost ambiental i amb millors resultats a llarg termini (Hobbs & Humphries 1995, Lodge et al. 2006, Hulme 2006). El paper essencial de la prevenció ha estat destacat en diverses estratègies de conservació, com el Conveni de Biodiversitat (http://www.biodiv.org/) i l'Estratègia Global d'Espècies Invasores (McNeely et al. 2001).

La prevenció inclou totes aquelles polítiques i mesures que una localitat o regió implementa per evitar l'entrada d'una espècie o per procedir al seu control immediat en cas que acabi d'entrar. Les actuacions per reduir el risc d'invasió d'una espècie es poden realitzar abans de la seva arribada (al país d'origen o en la via d'entrada), a l'arribar (a la zona d'intercepció) o com a reacció d'emergència a la seva detecció.

Un aspecte crític en la prevenció és la identificació d'espècies potencialment invasores, la qual sol ser difícil per diverses raons. En primer lloc, és difícil prevenir l'entrada d'espècies potencialment invasores ja que predir sense cap metodologia quines de les espècies introduïdes podran establir poblacions viables i es propagaran no és feina fàcil. En segon lloc, els impactes d'espècies exòtiques potencialment invasores no sempre són coneguts i a més, poden variar entre ecosistemes i entre regions (Erhenfeld 2010). Per tant, és necessari establir protocols, el més acurats possible, que permetin preveure el potencial invasor de les espècies introduïdes (Verbrugge et al. 2010).

El coneixement científic acumulat sobre els trets biològics de les especies invasores (capacitat invasora), les característiques dels hàbitats envaïts (invasibilitat) i els altres factors d'influència esmentats (vegi's l'apartat Factors que determinen l'èxit d'invasió) han estat el fonament per desenvolupar esquemes d'**avaluació de risc** que, actualment, constitueixen una de les eines de predicció i prevenció d'invasions més importants (Gordon et al. 2008). Alguns d'aquests esquemes intenten identificar la fracció d'espècies introduïdes amb una alta probabilitat d'esdevenir invasores per evitar que s'expandeixin i causin impactes (Andersen et al. 2004). La utilització d'aquests esquemes com a eina de gestió pot ajudar als gestors ambientals a justificar la prohibició d'entrada de determinades plantes exòtiques així com a establir prioritats en l'aprofitament del temps i dels recursos en cas de comptar amb diverses espècies potencialment invasores. Aquestes avaluacions també poden ajudar a obtenir i reforçar el suport del públic, imprescindible per a una gestió efectiva. La implementació d'aquest tipus de protocols d'avaluació de risc a altres països demostra que el seu ús produeix beneficis econòmics nets. És el cas d'Austràlia, on s'ha quantificat que l'ús d'aquests protocols ja ha estat beneficiós durant l'última dècada i es preveu que estalviarà a l'Estat més d'1,8 bilions al llarg dels propers 50 anys (Keller et al. 2007).

A Espanya s'han portat a terme bastants estudis sobre la capacitat invasora d'algunes espècies en certs hàbitats, com per exemple *Cortaderia selloana* al litoral català (Domènech et al. 2005), *Oxalis pes-caprae* a les Balears i a la costa llevantina (Gimeno et al. 2006), o *Ailanthus altissima* i *Carpobrotus sp*. a diverses illes (Traveset et al. 2008). Paral·lelament, altres estudis han determinat quins hàbitats i regions són més envaïts (Pino et al. 2005, Vilà et al. 2007, Gassó et al. 2009a). Recentment, també s'ha testat amb èxit a Espanya l'eficàcia de l'anàlisi de risc utilitzat a Austràlia per plantes exòtiques (*Australian Weed Risk Assessment;* Gassó et al. 2009b). Malgrat això, no s'han identificat, encara, espècies que podrien ser potencialment invasores al nostre territori si s'introduïssin al medi natural, per tal de poder prevenir la seva entrada i evitar possibles impactes.

### *Detecció ràpida i eradicació*

L'eradicació definitiva d'una espècie invasora només acostuma a ser viable durant les primeres fases d'introducció o en poblacions aïllades (Myers et al. 2000, Zavaleta et al. 2001, Groves & Panetta 2002). Una detecció i resposta ràpida implica la realització d'inversions per tal d'identificar i gestionar ràpidament espècies exòtiques de recent introducció, o amb poblacions molt localitzades (Hulme et al. 2009). La identificació passiva d'aquestes espècies pot resultar en un retard significatiu entre la introducció i el descobriment (Costello & Solow, 2003), temps durant el qual l'espècie pot establir-se i expandir-se a diverses àrees. No obstant això, la recerca activa i la identificació ràpida de nous invasors pot ser un procés difícil i molt costós (Horan & Lupi 2010). Per tant, la disponibilitat d'una quantitat adequada de recursos econòmics durant un temps suficientment llarg i el suport social i polític a aquestes mesures són essencials (Mack et al. 2000).

## *Contenció i control*

Una vegada l'espècie invasora s'ha establert al nou territori, les opcions més adequades i efectives per tal de controlar la seva expansió passen o bé, per limitar la seva àrea de distribució, el que es coneix com a contenció, o bé per reduir la seva densitat sense que això impliqui necessàriament disminuir l'àrea de distribució, és a dir, un control poblacional (Mack et al. 2000, Rejmánek 2000, Genovesi & Shine 2004, Hulme 2006, Clout & Williams 2009). Els mètodes variaran en cada cas, en funció de les característiques de les espècies invasores, les característiques de la zona envaïda i els objectius de gestió perseguits (Genovesi & Shine 2004, Dana & Rodríguez-Luengo 2008).

Els mètodes de contenció o control disponibles per fer front a les espècies invasores es classifiquen en quatre grans categories: manuals, mecànics, químics i biològics. Aquests mètodes es poden utilitzar de forma individual o combinats (Mack et al. 2000). Els mètodes manuals normalment s'apliquen en zones reduïdes i/o susceptibles a ser malmeses per actuacions mecàniques o químiques, les quals es basen en l'ús de maquinària o productes químics per eliminar extensions grans envaïdes. El control mecànic si bé pot ser efectiu, normalment no és pràctic en àrees molt extenses i a més pot crear fortes pertorbacions a l'hàbitat gestionat. El control químic és probablement la eina principal en el combat de les espècies exòtiques en l'agricultura. Malauradament, aquest mètode sovint comporta riscos tant per a la salut dels éssers humans com per moltes altres espècies natives que no són objectiu de control. A més, moltes de les espècies invasores més problemàtiques són aquàtiques o de ribera, hàbitats on l'ús de substàncies químiques pot tenir efectes devastadors. L'elevat cost d'aquestes mesures i la necessitat d'aplicacions repetides fa que moltes vegades sigui impossible la continuïtat d'aquests mètodes.

Per altre banda, el control biològic utilitza organismes vius, normalment també introduïts, que ataquen a l'espècie que es vol controlar. Normalment s'utilitzen enemics naturals de l'espècie invasora, que poden ser fitòfags, depredadors o patògens. Es tracta d'una estratègia a llarg termini que s'utilitza per reduir la densitat de les poblacions establertes en casos on són impossibles d'eradicar (Sheppard et al. 2006). Aquesta estratègia de control és força utilitzada a països com Nord Amèrica o Sud-àfrica, però dins de la Comunitat Europea no s'ha alliberat mai un agent de control per tal de controlar cap planta invasora (Shaw 2003). Entre d'altres motius perquè és un mètode força criticat atès que, a vegades, els agents de control introduïts poden tenir efectes perjudicials sobre les espècies natives i perquè ells mateixos poden acabar convertint-se en noves espècies invasores.

Per tant, donat que tots aquests mètodes tenen els seus avantatges i inconvenients, han de ser avaluats acuradament per tal d'identificar el més adient en cada situació o valorar la seva acció combinada.

### *Manteniment, restauració i seguiment*

El manteniment i la restauració dels ecosistemes en els que s'ha eliminat una espècie invasora i el seu posterior seguiment, tenen una influència considerable en l'èxit de les actuacions a llarg termini, sobretot en el cas de les plantes invasores. Les actuacions de manteniment, per una banda, eviten la reaparició de l'espècie invasora i d'altres espècies exòtiques que puguin aprofitar el buit ecològic per instal·lar-se. Per altre banda, les actuacions de restauració estan encaminades a restablir l'estructura de la comunitat nativa i el funcionament de l'ecosistema, i a facilitar la recuperació de les espècies autòctones. La necessitat d'aquests programes depèn tant de les característiques de l'espècie invasora i de l'estadi de la invasió, com dels impactes que l'espècie hagi pogut causar sobre l'ecosistema receptor.

L'eradicació o el control poblacional d'una espècie exòtica vegetal establerta al territori rarament s'aconsegueix amb actuacions puntuals, ja que moltes espècies invasores tenen bancs de llavors o rizomes molt persistents que fan que es necessitin actuacions repetides durant diversos anys consecutius per tal d'eliminar-les (D'Antonio & Meyerson 2002). A més, una de les conseqüències de l'eliminació d'espècies invasores, en el cas de les plantes, és la facilitació de la proliferació d'altres espècies exòtiques (Álvarez & Cushman 2002, Ogden & Rejmánek 2005, Hulme & Bremner 2006, Truscott et al. 2008), o bé la pertorbació del sòl i de la vegetació circumdant (D'Antonio et al. 1998, Zavaleta et al. 2001). Per tant, sense activitats de manteniment a llarg termini, l'eradicació o control d'una determinada espècie invasora acaba normalment sent poc efectiva (Briggs & Cornelius 1998, Dana & Rodríguez-Luengo 2008, Shafroth et al. 2008). També, cal tenir present que moltes vegades la simple eliminació d'una espècie invasora que està causant un impacte en un indret determinat, no és suficient per a que la comunitat nativa es recuperi (Simberloff 2003). Diversos factors poden influenciar de manera important els resultats de les actuacions d'eradicació o control, que, finalment, podrien no assolir els nivells de recuperació de la comunitat nativa desitjats (Partel et al. 1998, Zavaleta et al.

2001). Aquests factors poden ser, entre d'altres, els canvis d'usos del sòl històrics, la disponibilitat de banc de llavors d'espècies autòctones, l'aparició d'altres espècies exòtiques o el règim de pertorbacions.

El seguiment després de les mesures de control o restauració, tant de la resposta de les espècies invasores tractades com de les espècies autòctones és extremadament valuós per assegurar l'èxit de l'actuació, ja que permet controlar l'aparició d'efectes secundaris indesitjats, determinar la durada de les actuacions de manteniment i finalment, avaluar la necessitat d'aplicar mesures de restauració específiques (Zavaleta et al. 2001, Wotton et al. 2004, Dana & Rodríguez-Luengo 2008). Finalment, l'ús d'indicadors d'eficàcia durant aquest seguiment i l'establiment de zones de referència sense espècies exòtiques, són imprescindibles per assegurar una correcte gestió i proporcionar eines als gestors per fer un seguiment de l'evolució de les seves actuacions (Chapman & Underwood 2000, D'Antonio et al. 2004). La comparació entre les zones on s'ha eliminat una espècie invasora i les zones intactes de referència, ens permet determinar si la vegetació nativa cada vegada s'assembla més a la vegetació de les zones control o de referència (Chapman & Underwood 2000, Mason & French 2007).

Tot i la importància demostrada del manteniment, la restauració i el seguiment moltes vegades no es duen a terme ja que els programes de control son dissenyats normalment a curt termini (Hobbs & Humphries 1995). No obstant, donat que les mesures de gestió són extremadament costoses tant en recursos humans i tècnics com econòmics (Pimentel et al. 2005) és important assegurar la seva eficàcia. Per aquest motiu, en aquesta tesi hi ha dos capítols dedicats al seguiment i l'avaluació de les mesures d'eradicació o control de plantes invasores.

## Marc legislatiu per a la gestió de les invasions biològiques

Davant de la problemàtica de les espècies invasores, diversos organismes internacionals, governs i ONGs ja van reconèixer a la Convenció de Biodiversitat (CBD 1992) la necessitat de "prevenir la introducció, i controlar o eradicar les espècies invasores ja establertes que amenacessin els ecosistemes, els hàbitats o les espècies natives" (Article 8 h, 5 juny 1992; http://www.biodiv.org). En l'àmbit europeu, s'està desenvolupant l'Estratègia sobre Espècies Exòtiques Invasores (Genovesi & Shine 2004), que constitueix una eina important per implementar els instruments legals vigents (Conveni de Berna, Directives de la CEE, etc.), i promoure el desenvolupament dels plans d'actuació en cada un dels estats membres.

A Espanya, el *Ministerio de Medio Ambiente*, seguint les recomanacions de la Convenció de Biodiversitat va redactar el 1998 la *Estrategia española para la conservación y el uso sostenible de la biodiversidad* en la que es proposava l'elaboració i harmonització dels recursos jurídics i tècnics necessaris per controlar, i/o impedir la introducció d'espècies exòtiques que amenacessin els sistemes, hàbitats, espècies o poblacions autòctones (http://www.mma.es/conservnat/planes.htm). A més, el desembre del 2007 es va publicar la Llei 42/2001 del Patrimoni Natural i de la Biodiversitat («BOE» 299, de 14-12-2007) (http://www.boe.es/boe/dias/2007/12/14/pdfs/A51275-51327.pdf) on hi ha un article (Llei 42, capítol III, article 61) dedicat a la prevenció i control d'espècies exòtiques, i on es proposa la creació d'un Catàleg Espanyol d'Espècies Exòtiques Invasores. En aquest catàleg s'hauran d'incloure totes aquelles espècies exòtiques invasores que constitueixin una amenaça greu per a les espècies autòctones, els hàbitats o els ecosistemes, l'agronomia o per als recursos econòmics associats a l'ús del patrimoni natural. Aquest article especifica que la inclusió en aquest catàleg d'espècies exòtiques comporta la prohibició genèrica de possessió, transport, tràfic i comerç d'exemplars vius o morts, de les seves restes o propàguls, incloent el comerç exterior, d'aquestes espècies. L'elaboració d'aquest catàleg nacional està actualment en procés. Finalment, l'article insta a cada Comunitat Autònoma a dur a terme un seguiment i un control de les espècies exòtiques presents en els seus territoris, i a desenvolupar catàlegs autonòmics i estratègies de gestió específiques. En aquesta tesi ens hem esmerçat en fer una recerca aplicada a la confecció de catàlegs de plantes exòtiques.

## Objectius i estructura de la tesi

Donada la necessitat d'aplicar mesures adequades per a la gestió d'espècies exòtiques, i d'integrar aquestes mesures en les estratègies de conservació de la biodiversitat, les diferents administracions autonòmiques espanyoles han dut a terme actuacions de gestió d'espècies invasores. No obstant això, la informació sobre aquesta gestió està bastant dispersa i manca comunicació entre administracions. Abans d'aquesta tesi, no existia cap avaluació de les mesures de gestió dutes a terme, ni de la percepció dels gestors sobre les invasions biològiques, ni sobre quines són les limitacions amb les que es troben a l'hora de fer front a aquesta problemàtica. A més, normalment no hi ha gaire flux d'informació entre el món acadèmic i el món de la gestió. Entre d'altres motius perquè la informació acadèmica sobre la gestió d'espècies exòtiques és insuficient i de difícil accés pels gestors.

Per aquests motius, el principal objectiu d'aquesta tesi ha estat aprofundir en les mesures de gestió de les plantes exòtiques a Espanya (**Capítol 2**), identificant quines són les principals limitacions d'aquestes mesures. Conèixer aquestes limitacions ens ha dut a plantejar-nos els següents objectius específics:

i) Determinar les principals espècies vegetals problemàtiques a Espanya i avaluar els criteris que s'utilitzen a l'hora de prioritzar la seva gestió (**Capítol 2**).

ii) Identificar i classificar les espècies potencialment invasores per Espanya **(Capítol 3).**

iii) A nivell global, quantificar els impactes de les plantes invasores i les conseqüències de la seva eliminació en la riquesa i abundància d'espècies natives **(Capítol 4).**

iv) A nivell regional i com a cas d'estudi, avaluar l'eficàcia de l'eliminació manual de *Carpobrotus sp.* a la costa d'Andalusia, i la recuperació de la vegetació nativa **(Capítol 5).**

En el **Capítol 2** s'ha recopilat tota la informació disponible sobre les mesures de gestió de plantes exòtiques dutes a terme a Espanya en els últims 10 anys. La informació s'ha obtingut a través de qüestionaris enviats als gestors ambientals encarregats de la conservació del medi natural a cada una de les administracions autonòmiques espanyoles. Amb la informació obtinguda (1) s'ha avaluat la percepció per part dels gestors de la problemàtica de les invasions per plantes exòtiques, (2) s'ha identificat l'estatus, la distribució i els principals impactes de les plantes exòtiques gestionades a Espanya, (3) s'ha estudiat el tipus de gestió dut a terme i els seus costos associats i, finalment (4) s'han avaluat els criteris que els gestors utilitzen per prioritzar la gestió de les plantes exòtiques a Espanya (Hiebert 1997). Aquest primer capítol ha permès identificar les principals limitacions de les mesures de gestió aplicades a Espanya i ha donat peu als capítols següents.

El **Capítol 3** es centra en la importància de la prevenció per tal de fer front al problema de les plantes invasores. A Espanya un dels àmbits on s'ha avançat menys pel que fa a les espècies exòtiques és en la posada a punt d'estratègies de prevenció que incorporin metodologies d'anàlisi de risc. Aquest capítol identifica les plantes exòtiques que podrien esdevenir invasores a Espanya si s'introduïssin al medi natural. Per fer això, en primer lloc s'han identificat les plantes amb comportament invasor a regions veïnes o regions de clima mediterrani d'arreu del món. Després, mitjançant l'ús d'un protocol d'avaluació de risc desenvolupat per Austràlia (*Australian Weed Risk Assessment;* Pheloung et al. 1999) i d'un altre desenvolupat per Europa Central (Weber & Gut 2004), aquestes espècies s'han classificat segons la seva capacitat invasora.

Els capítols 4 i 5 fan referència a la necessitat de dur a terme un seguiment un cop acabades les actuacions inicials d'eradicació i control de plantes exòtiques, per tal d'assegurar l'eficàcia d'aquesta gestió i avaluar si es necessiten mesures de restauració específiques.

Al **Capítol 4** s'ha realitzat una revisió bibliogràfica global per tal d'avaluar els impactes de les plantes invasores i les conseqüències de la seva eliminació en la vegetació nativa. Mitjançant un metanàlisi s'ha quantificat la magnitud i la direcció del canvi en la riquesa i en l'abundància d'espècies natives en la vegetació envaïda i un cop l'espècie invasora s'ha eliminat. S'ha determinat si (1) les invasions per plantes exòtiques provoquen una davallada de la riquesa i abundància de plantes natives, (2) els indrets on s'han eliminat les plantes exòtiques són similars als indrets de referència (no envaïts i no tractats) i, finalment, (3) s'han identificat les comparacions utilitzades a l'hora de fer un seguiment i avaluació de les actuacions d'eliminació.

En el **Capítol 5** s'ha utilitzat com a cas d'estudi l'eliminació d'una de les plantes invasores més esteses i problemàtiques al litoral espanyol, el bàlsam (*Carpobrotus sp.*). S'ha avaluat l'eficàcia a curt termini de la seva eliminació, així com també la recuperació de la vegetació nativa en diferents ecosistemes de la costa d'Andalusia. En aquest estudi s'han comparat parcel·les envaïdes per *Carpobrotus,* parcel·les on *Carpobrotus* havia estat eliminat manualment i parcel·les de referència (o controls) no envaïdes i no tractades.

Per finalitzar amb la tesi, en el **Capítol 6** es sintetitzen i discuteixen els principals resultats dels quatre capítols precedents i les conclusions finals.

Els quatre capítols centrals es presenten en format d'article científic, la qual cosa comporta algunes redundàncies en les introduccions dels diferents capítols, però les referències i els annexes han estat agrupats al final de la tesi per evitar repeticions i facilitar la lectura. Els capítols centrals i la discussió general estan íntegrament escrits en anglès. Les conclusions finals de la tesi es presenten en català i en anglès tal com s'ha fet en aquesta introducció general.

# Chapter 1

# General introduction

## Biological invasions

Humans have transported species outside their native ranges, either intentionally or unintentionally, since immemorial times. However, in the last decades, factors such as ecological and economic globalization have contributed to dramatically accelerate the number of alien species that are introduced to a new region as well as the number of species that establish new populations in these regions (McNeely et al. 2001, Kowarik 2003, Levine & D'Antonio 2003). Some of these species not only get established but also spread, expanding through large areas in a short time (Pyšek et al. 2004). These species are called **invasive species**. During this expansion process, some of them can cause significant impacts, both environmental and socioeconomic.

Consequently, biological invasions are considered nowadays to be an important component of global change and one of the most serious threats to global biodiversity and ecosystems integrity (Vitousek et al. 1997, Mack et al. 2000, Sala et al. 2000, Brooks et al. 2004, Thuiller et al. 2007). Compelling evidences exists that the magnitude of this threat will continue to increase globally, as international trade and tourism continue to increase and, while climate and land use continue to change (Perrings et al. 2005, Hulme 2009, Walther et al. 2009, Vilà & Ibáñez 2011).

Among the ecological impacts caused by invasive species the homogenization of global biodiversity through the loss of local native diversity is one of the most important impacts. Several mechanisms such as competition, herbivory, predation, disease transmission, hybridization, etc., have been described to be responsible for native diversity loss (Mack et al. 2000, Sala et al. 2000, Blackburn et al. 2004, Gaertner et al. 2009, Kettunen et al. 2009). In addition, invasive species can also produce changes in the structure and functioning of ecosystems and the services they provide (Vitousek 1994, Charles & Dukes 2008, Hejda et al. 2009, Kettunen et al. 2009, Pejchar & Mooney 2009, Erhenfeld 2010, Vilà et al. 20010), through, for example, changes in disturbance regimes (D'Antonio & Vitousek 1992, D'Antonio et al. 1999, Brooks et al. 2004, Pauchard et al. 2008), the water cycle (Williams & Baruch 2000, Gerlach 2004, Potts et al. 2008) or biogeochemical cycles (Liao et al. 2008, Raizada et al. 2008, van Kleunen et al. 2010, Vilà et al. 2011).

The introduction of alien species not only generates ecological impacts, but also has economic and human welfare consequences (Soulé 1992, Born et al. 2005, Pimentel et al. 2005, Lovell et al. 2006, Olson 2006, Kettunen et al. 2009). The economic costs caused by these species can be extremely high, either due to direct losses in several economic sectors such as agriculture, fishing or navigation, or the subsequent costs arising from their management (Pimentel et al. 2005, Kettunen et al. 2009). Pimental et al. (2005) estimated that the economic damage associated with alien species impacts and emerging diseases together with their control costs in USA exceeds $120 billion per year. In Europe, according to a report by the European Commission (Kettunen et al. 2009), it has been estimated that this figure reaches at least the 12.5 billion Euros annually. In Spain, the invasion of water hyacinth (*Eicchornia crassipes*) in the Guadiana River deserves special attention due to the huge amount of money invested in its removal (~ € 7 million per year) and the significant damages caused in different sectors such as fishing or navigation.

In particular, invasive species can reduce crops and timber production, diminish the quality of pasture, affect communication routes, clog pipeline and irrigation infrastructures, reduce water availability and influence the recreational and aesthetic value of the natural areas invaded. Regarding the impacts on human health it is important to note that some illnesses such as allergies or dermatitis can be caused by alien species such as *Heracleum mantegazzianum, Ambrosia artemisiifolia* or *Cortaderia selloana* (Kettunen et al. 2009). Moreover, some alien species might be vectors of parasites or pathogens to humans or domestic animals, increasing the transmission of infectious diseases (Pimentel et al. 2005).

All these impacts are gaining great international significance and several countries are increasingly adopting management measures to minimize the negative consequences of invasive species.

## *The invasion process: definitions*

To manage biological invasions, it is important to understand first, the process by which an alien species becomes invasive. This is the result of a sequence of successive stages (Kolar & Lodge 2001, Leung et al. 2002), which are illustrated in Figure 1.1 for plants, which are the main study taxa in this thesis. The different stages of this process do not necessarily occur in all cases, since only few species will pass to the following stage (Kolar & Lodge 2001). Therefore, from the total number of species that is transported to a new region only a small percentage will be able to become invasive.

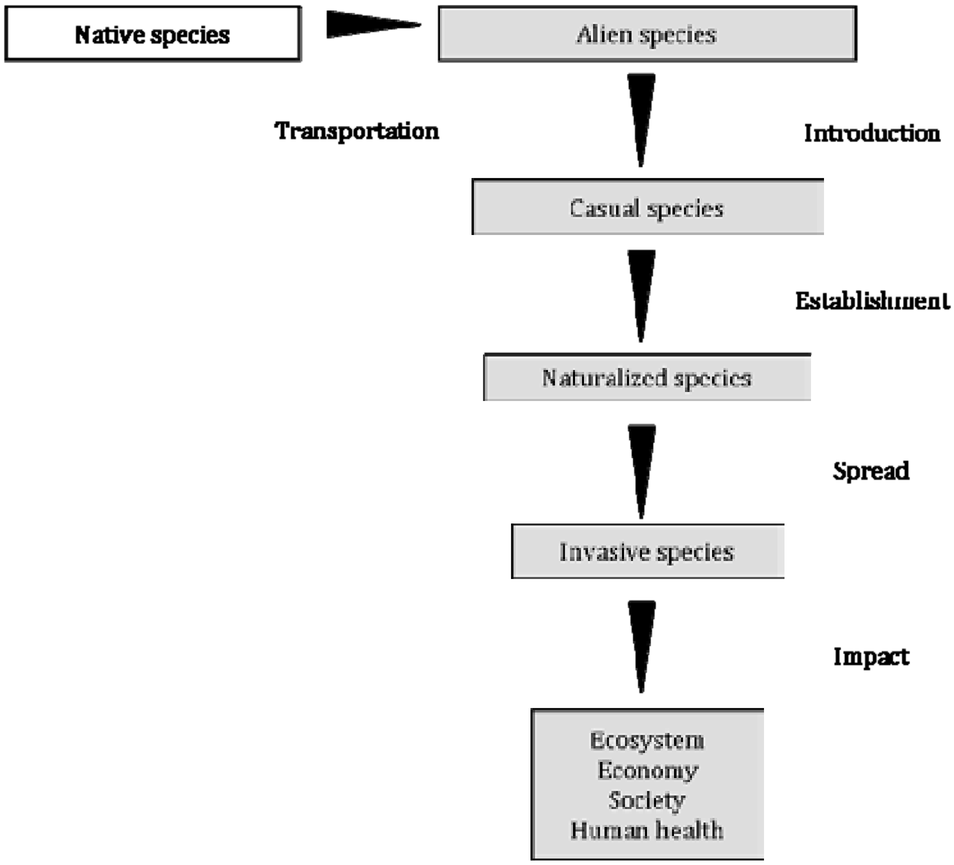



**Figure 1.1** Diagram of the invasion process adapted from Pyšek et al. (2004) and Duncan et al. (2003).

Here we define each of the stages of the invasion process:

**Transportation:** Species must be transported from their native area to a new region. In the new location the species is called **alien species** (*Synonyms:* non-native, introduced, allochthonous, foreign). An alien species is a species present in a given region (e.g., continent, island, eco-region, or any political entity) due to human intervention (Pyšek et al. 2004), or that has arrived without the help of people from an area in which it was alien (Hulme et al. 2008b). Introduction pathways of these species are diverse but can be divided simply, in intentional or unintentional:

- **Intentional introduction:** the species is introduced with a particular purpose either legally or illegally. For example, plants can be introduced for their use in agriculture, forestry, as ornamental species, for soil protection, for obtaining medicines, fibers or raw materials for industry, etc.

- **Unintentional introduction:** the species is introduced unintentionally by humans through, for example, the trade of a commodity (e.g., wheels of trucks) or as a contaminant of agricultural products (Hulme 2005, Hulme et al. 2008b).

**Introduction:** The species should be introduced to the natural environment of the new region. If it comes from an unintentional release, the distinction between transportation and introduction is virtually impossible. However, if it comes from an intentional release, the introduction pathway to the natural environment can be either deliberate (e.g., dumping of garden waste) or accidental (e.g., dispersal of seeds from the gardens; Hulme et al. 2008b). Once the species are introduced into the natural environment they are called **casual species** (*Synonyms:* not established). Casual species are alien species that may reproduce occasionally outside cultivation or captivity in an area, but that eventually die out because they do not form self-sustaining populations, and rely on repeated introductions for their persistence (Pyšek et al. 2004).

**Establishment:** Following introduction, the species can get established, becoming **naturalized** or **established species**, which are alien species that sustain self-replacing populations for at least 10 years without direct human intervention (Pyšek et al. 2004).

**Spread:** The species that establish successfully may increase in abundance and spread beyond their point of release. These are considered **invasive species**. In the literature the use of this term is controversial. There are, mainly two points of view when defining invasive species. One definition is based on the expansion capacity and the other on the ability of the species to cause impacts. The first definition, mostly used by ecologists, considers that invasive species are naturalized species that produce reproductive offspring, often in very large numbers, at considerable distances from the parent individuals, and thus have the potential to spread over large areas (Pyšek et al. 2004). The other definition is more restrictive, as it considers invasive species as naturalized species that produce significant changes in the composition, structure or functioning of ecosystems. This definition is used by national organizations as GAE (Group of exotic birds) or GEIB (Specialist Group in Biological Invasions), and by international institutions such as the International Union for Conservation of Nature (IUCN; http:// www.issg.org/) or the Convention on Biological Diversity (CBD; http://www.cbd.int/). In this thesis we adopted the first definition, which is based on geographic criteria, because we believe that to consider the environmental impact in order to decide whether a species is invasive or not, is not always useful. Very often, the real impact is unknown because it has not been quantified or because it is difficult to estimate (Vilà et al. 2009).

### *Factors determining invasion success*

Predicting the success of alien species consist in determining the **risk of invasion**. This has been a major aim of ecological research since invasions were recognized as an important conservation issue (Elton 1958, Reichard & Hamilton 1997, Lonsdale 1999). Invasion success depends on the success of particular invasive species and the level of invasion (i.e., alien species abundance and richness in the region of introduction). Several non-exclusive factors have been suggested to be related with invasion success (Figure 1.2).

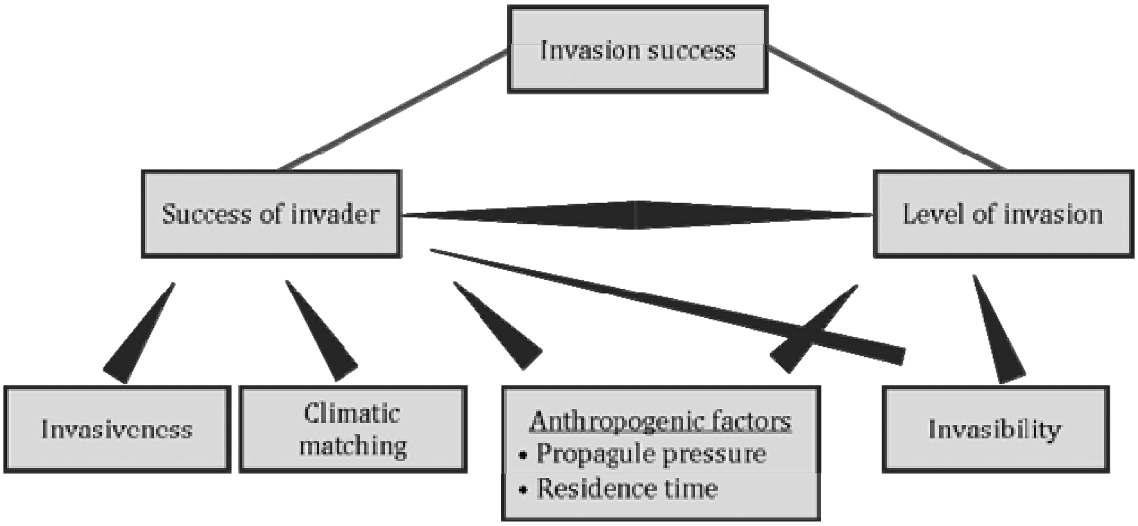



**Figure 1.2** Diagram of the components of successful invasion and its influence factors. Modified from Gassó (2008).

On the one hand, the success of invaders is mainly influenced by invasiveness and climate matching. **Invasiveness** or the **invasion potential** of species is the extent to which a species per se is able to overcome various biotic and abiotic barriers and become invasive (di Castri 1989). Many studies have attempted to profile successful invaders features. However, several studies have found that identifying traits consistently associated with invasiveness is difficult (Pyšek & Richardson 2007). Some of the biological and ecological characteristics of invasive species identified so far are: high competitiveness, rapid growth, high fecundity, efficient dispersal, small genome, and high specific leaf area, among others (Rejmánek 1996, Rejmánek & Richardson 1996, Lake & Leishman 2003).

The other main environmental factor influencing the success of a particular invader is climate. The **climate matching** hypothesis states that species should have a better chance of establishing if climate at the location of introduction and at the species' natural range is closely matched (Brown 1989, Panetta & Mitchell 1991, Scott & Panetta 1993, Williamson 1996). Therefore, species of similar latitude or biogeographic regions will be more successful invaders (Sun et al. 2005). There are several studies that support this hypothesis (Curnutt 2000, Blackburn & Duncan 2001b, Cassey 2003, Hayes & Barry 2008), and even use it as a basis for risk invasion models (Thuiller et al. 2005).

On the other hand, variations in the level of invasion between localities are also dependent on differences in **invasibility**. Invasibility is the resistance that the recipient ecosystem offers to invasion (Figure 1.2). To determine if an ecosystem is more or less prone to invasion, we must clarify what abiotic and biotic factors regulate seed germination and seedlings establishment at that ecosystem (Lonsdale 1999). Species diversity, interspecific interactions (mutualism, competition, herbivory), disturbance and soil nutrient availability have been the main factors examined to detect differences in invasibility. It has been suggested that eutrophication and the availability of bare soil are conditions that offer greater invasibility to an ecosystem (Kowarik 1990, Burke & Grime 1996, Lonsdale 1999, Hobbs 2000).

Finally, **propagule pressure** and **residence time** are the other two factors that also determine the success of invasion (Blackburn & Duncan 2001a, Duncan et al. 2003; Figure 1.2). Propagule pressure or **introduction effort** (Blackburn & Duncan 2001a) is a composite measure of the number of individuals released into a region where they are not native (Carlton 1996). Residence time (i.e., time since introduction) integrates aspects of propagule pressure related to the timing of invasion: the longer the species is present in the region, the greater the size of the propagule bank, and the greater the probability of dispersal and establishment of new populations (Rejmánek et al. 2005b). The ecological impact of introduced species might also increases with time of residence (Collier et al. 2002).

## Management of biological invasions

Now that the increasing impacts and costs of invasive species are being recognized management of invasive species has become an important challenge and a high priority for environmental managers, especially in protected areas. The possible approaches for alien plant management include three stages, which are complementary: 1) prevention, 2) early detection and eradication, 3) containment and control measures, accompanied by measures of impacts mitigation (Rejmánek 2000, Hulme 2006, NISC 2008). The maintenance, restoration and monitoring of the ecological integrity of the managed ecosystem, despite not being considered management methods in themselves, are clearly necessary once you have carried out measures for the eradication, containment or control of a particular invasive species.

The adoption of one or another management strategy will depend mostly on the stage of the invasion process (Figure 1.3) and the real possibilities of management success according to the characteristics of the species and the managed habitat, the available resources, the institutional support and the distribution of management efforts over time (Hulme 2006).

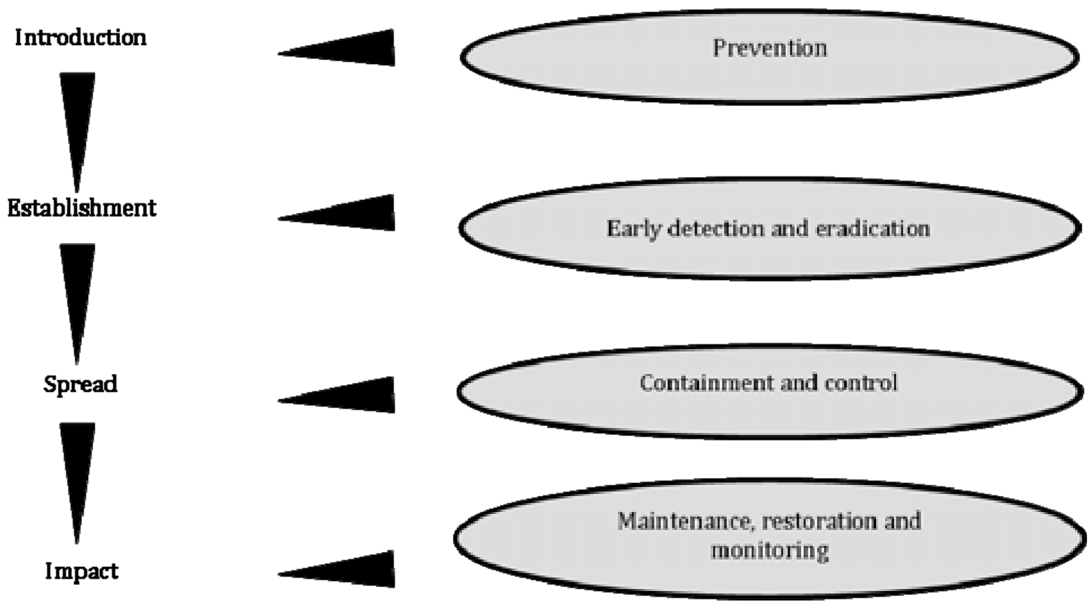



**Figure 1.3** Relationship between the different stages of invasion and the possible strategies used to deal with biological invasions. Diagram adapted from Hulme (2006).

## *Prevention*

Prevention measures aim to avoid the entry of potentially invasive species into a new region. Given that in general control or eradication strategies are difficult and very costly prevention should be the first strategy to cope with biological invasions (Hulme 2006). Prevention is widely promoted as being by far the most effective and environmentally desired strategy, and with better long-term results than the other management actions (Hobbs & Humphries 1995, Lodge et al. 2006, Hulme 2006). The role of prevention has been highlighted by recent conservation strategies, such as the Convention on Biodiversity (http://www.biodiv.org/) and the Global Strategy on Invasive Alien Species (http://www.gisinetwork.org/Documents/globalstrategy.pdf; McNeely et al. 2001).

Prevention includes all those policies and measures that a region must implement to prevent the entry of an invasive species or to control it immediately after its entry. Actions to reduce the risk of invasion of an alien species can be made before its arrival (at the country of origin or at the pathway of introduction), when just arrived (in the area of interception) or as an emergency response when it is first detected.

An important aspect of prevention is the identification of potentially invasive species. This is often difficult for several reasons. First, it is difficult to predict which of the introduced species will establish viable populations and will propagate without an effective methodology. Second, the impacts of potentially invasive species are not always known and may also vary among ecosystems and regions (Erhenfeld 2010). It is therefore necessary to establish protocols as accurate as possible in order to predict the invasive potential of introduced species (Verbrugge et al. 2010).

The accumulated scientific knowledge on invader traits (invasiveness) together with the characteristics of invaded habitats (invasibility), and the other influencing factors already mentioned (see Factors determining invasion success), has been the basis for developing **risk assessment** schemes that currently constitute one of the most important tools for predicting and preventing (Gordon et al. 2008). These schemes attempt to identify the fraction of introduced species with a high likelihood of becoming invasive, and prevent their spread and their damaging effects (Andersen et al. 2004). The use of these schemes as a management tool can help environmental managers to justify the entry prohibition of potentially invasive plants and to set priorities for management when several potentially invasive species are present. Risk assessments can also help to strengthen and gain public support, which is essential for an effective management. So far, there is evidence that the implementation of risk assessment protocols produces net economic benefits. This is the case of Australia, where it have been quantified that the use of these protocols has already been beneficial over the last decade, and it is expected to save the State more than 1, 8 billion over the next 50 years (Keller et al. 2007).

In Spain, several studies have investigated the invasiveness of particular species in certain habitats, such as *Cortaderia selloana* in the Catalan coast (Domènech et al. 2005), *Oxalis pes-caprae* in the Balearic Islands and the Levantine coast (Gimeno et al. 2006), and *Ailanthus altissima* and *Carpobrotus sp.* in several Mediterranean islands (Traveset et al. 2008). Similarly, other studies have determined which habitats and regions are more invaded (Pino et al. 2005, Vilà et al. 2007 Gassó et al. 2009a). Recently, a study has also tested successfully for Spain the effectiveness of the risk assessment developed in Australia (Australian Weed Risk Assessment; Gassó et al. 2009b). However, potentially invasive species not yet introduced to the natural systems in Spain have not been identified, despite that this is essential in order to prevent their entry and avoid future impacts.

### *Early detection and eradication*

The complete eradication of invasive species is usually only feasible during the early stages of introduction or in isolated populations (Myers et al. 2000, Zavaleta et al. 2001, Panetta & Groves 2002). Early detection and rapid response involve making investments to identify and address newly introduced, localized populations of invasive species (Hulme et al. 2009). Passive discovery of invasive species can result in a significant time lag between introduction and discovery (Costello & Solow 2003), during which time the invader can gain a significant foothold and spread to multiple areas. However, it may be difficult and costly to actively search for and discover a newly introduced invader (Horan & Lupi 2010). Therefore, the availability of economic resources for a long-term management and the social and political support to early detection and eradication measures are crucial (Mack et al. 2000).

### *Containment and control*

Once an invasive species has established in a new region, the most appropriate and effective options to control its spread are containment and population control.

Containment limits the distributional range of the species and control reduces its density without necessarily diminishing its range of expansion (Mack et al. 2000, Rejmánek 2000, Genovese & Shine 2004, Hulme 2006, Clout & Williams 2009). The methods used vary, depending on the characteristics of invasive species, the characteristics of the invaded area and the management objectives pursued (Genovese & Shine 2004, Dana & Rodríguez-Luengo 2008).

Containment or control methods are classified into four broad categories: manual, mechanical, chemical and biological control. These methods can be used individually or in combination (Mack et al. 2000). Manual methods are typically applied in small areas or in areas that are likely to be damaged by mechanical or chemical actions, which are based on the use of machinery or chemicals to remove large invaded extensions. Although mechanical methods can be very effective, they may not be feasible in very large areas. Moreover, they can severely disturb the managed habitat. Chemical methods are probably the main tool in combating invasive species in agriculture. Unfortunately, chemical methods often involve risks, both for human health and for many non-target native species. In addition, many of the most problematic alien species invade aquatic or riparian habitats, where the use of chemicals may have devastating effects. The high costs of these measures and the need for repeated applications often make impossible the long-term viability of these control methods.

Biological control uses living organisms, usually also introduced, which attack the species to be controlled. This method generally employs natural enemies of the invasive species that can be phytophagous, predators or pathogens. It is a long-term strategy that aims to reduce the density of the established populations in those cases where the invasive species is impossible to eradicate (Sheppard et al. 2006). This control strategy is commonly used in countries such as North America or South Africa, but no classical biological control agent has ever been released against invasive plants within the European Union (Shaw 2003). Biological control has been highly criticized because sometimes the introduced control agents can have detrimental effects on native species and because they can become new invasive species in the regions where they have been introduced.

Therefore, given that all these methods have their advantages and disadvantages, they should be carefully evaluated in order to identify the most appropriate one in each situation and also to evaluate their combined effects.

### *Maintenance, restoration and monitoring*

Maintenance and restoration of ecosystems where a particular invasive species has been removed, and the subsequent monitoring have considerable influence on the success of the long-term management measures, particularly in the case of invasive plants. While maintenance activities prevent the re-establishment of the managed invasive species and other alien species, restoration measures try to restore the community structure and ecosystem functioning, and to facilitate the recovery of native species. The need of these programs depends on the characteristics of the invasive species, the stage of the invasion process, and on the impacts that the species may have already caused.

The eradication or control of alien plant species established in a territory is rarely achieved through short-term and punctual measures, because many species have extensive seed banks or persistent rhizomes which require repeated follow-up treatments, sometimes for several consecutive years, to eliminate them (D'Antonio & Meyerson 2002). Moreover, one of the consequences of invasive species removal may be to facilitate the proliferation of other alien species (Alvarez & Cushman 2002, Ogden & Rejmánek 2005, Hulme & Bremner 2006, Truscott et al. 2008), and to cause soil and vegetation disturbance (D'Antonio et al. 1998, Zavaleta et al. 2001).

In fact, many studies have suggested that in the absence of long-term maintenance activities, management measures commonly fail (Briggs & Cornelius 1998, Dana & Rodríguez-Luengo 2008, Shafroth et al. 2008). Moreover, the removal of invasive species alone does not always lead to the reestablishment of the desirable native communities (Simberloff 2003). Several factors can strongly influence the result of the restoration effort, which might not accomplish the desired level of recovery towards the native community characteristics (Partel et al. 1998, Zavaleta et al. 2001). These factors may be, among others, land use history, native seed bank availability, disturbance regime and the proliferation of other alien species.

Post-monitoring, on both the target invasive species and the invaded community, is extremely valuable in order to assure the success of the management measures. Monitoring may control unwanted side effects, may determine the length of the maintenance effort and, finally, can evaluate whether active restoration measures are needed or not (Zavaleta et al. 2001, Wotton et al. 2004, Dana & Rodríguez-Luengo 2008). The use of performance indicators during post-monitoring and the identification of reference non-invaded sites are essential to ensure proper management and provide managers with tools to document the outcome of the management (Chapman & Underwood 2000, D'Antonio et al. 2004, James et al. 2010). Comparisons between sites where an invasive species has been removed and intact reference areas, allow evaluating whether native plant community in removal sites is becoming similar to the native vegetation in species diversity and composition (Underwood & Chapman 2000, Mason & French 2007).

Despite the importance of maintenance, restoration and monitoring measures they are often forgotten because control programs are usually designed with short-term goals in mind (Hobbs & Humphries 1995). However, given that management measures are extremely costly in terms of human, technical and economic resources (Pimentel et al. 2005) it is really important to guarantee their effectiveness. This thesis has two chapters devoted to the evaluation of control measures carried out to cope with invasive plants.

## Legislative framework for the management of biological invasions

Given the evidences of invasive species impacts, international organizations, governments and NGOs have expressed their concerns about this issue. The Convention of Biological Diversity (CBD 1992) states the need to "prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species" (Article 8 h, 5 June 1992; http://www.biodiv.org). In Europe, a strategy on invasive alien species is being developed (Genovese & Shine 2004), as an important tool to implement existing legal instruments (Bern Convention, EU directives, etc), and to promote the development of action plans in the member States.

In Spain, the Environment Ministry, following the recommendations of the Convention of Biological Diversity drafted in 1998 the "Spanish strategy for the conservation and the sustainable use of biodiversity". This strategy proposes the development and harmonization of legal and technical resources necessary to control and/or prevent the introduction of alien species that threaten ecosystems, habitats and native species (http://www.mma.es/conservnat/planes.htm). In addition, in December 2007 the Law 42/2001 of the Natural Heritage and Biodiversity was published ("BOE" 299 of 14-12-2007) (http://www.boe.es/boe/dias/ 2007/12/14/pdfs/A51275-51327.pdf). This law contains an article (Law 42, Chapter III, Article 61) dedicated to the prevention and control of alien species, in which it is proposed the creation of a national catalog of invasive species. This catalog would include all those invasive species that constitute a serious threat to native species, habitats, ecosystems, agronomy or economic resources associated with the use of natural heritage. The article specifies that the inclusion in the catalog of a particular alien species involves the general prohibition of possession, transportation, traffic and trade of living or dead specimens, their remains or propagules, including the foreign trade of these species. The development of this national catalog is currently in process. The article also urges each Autonomous Community to undertake monitoring and control of alien species present in their territories, and to develop regional catalogs of invasive species and specific management strategies. This thesis contributes to build these species lists for Spain.

## Thesis objectives and outline

Responding to the need of implementing appropriate measures for the management of alien species, and to integrate these measures into conservation strategies, several Spanish administrative regions have carried out actions to cope with invasive species. Nonetheless, management information is quite scattered and there is a lack of communication between the different local, regional and national administrations. Prior to this thesis, there was neither an assessment of the management measures implemented in Spain nor an evaluation of managers' perception on biological invasions and of the limitations and impediments they encounter when dealing with this problem. Besides, usually there is a gap of information between academic research and environmental managers. Many times this is because the academic information on the management of alien species is insufficient and difficult to access by managers.

For these reasons, the main objective of this thesis is to investigate the management measures of alien plants carried out in Spain (**Chapter 2**) and to identify which are the main limitations of these measures. Once knowing these limitations the following specific objectives have been addressed:

i) To identify the most problematic invasive plant species in Spain and to assess the main criteria used to prioritize their management (**Chapter 2**).

ii) To identify and classify potentially invasive species in Spain (**Chapter 3**).

iii) To quantify, globally, the impacts of invasive plants and the consequences of their removal in terms of native species richness and abundance (**Chapter 4**).

iv) As a case study, to evaluate the efficacy of the manual removal of *Carpobrotus sp.* in the coast of Andalucía, and the recovery of the native vegetation after the alien plant removal (**Chapter 5**).

In **Chapter 2** all available information on management measures of alien plants in Spain in the last 10 years has been collected. Questionnaire surveys have been sent to managers responsible for nature conservation in each of the administrative regions of Spain. With the gathered information we have: (1) assessed human perceptions of alien species problems, (2) identified the status, occurrence and perceived impacts of noxious alien plants in natural areas, (3) described the management activities undertaken in order to prevent or control noxious alien plants and their associated costs, and, finally, we have (4) evaluated the criteria managers use to set priorities for management (Hiebert 1997). This chapter has identified the main limitations of management measures applied in Spain, which have gave rise to the following chapters.

**Chapter 3** focuses on the importance of prevention in order to successfully address the problem of invasive plants. In Spain one of the research areas where less progress has been made with regard to alien species is in the developing of prevention strategies that incorporate risk analysis methodologies. This chapter identifies the alien plants that may become invasive if introduced to the natural environment in Spain. First, we identified plants with invasive behavior in neighboring regions or Mediterranean climate regions around the world. Then, using a risk assessment protocol developed for Australia (Australian Weed Risk Assessment; Pheloung et al. 1999), and another developed for Central Europe (Weber & Gut 2004), these species have been ranked according to their invasive potential.

Next, chapters 4 and 5 refer to the need to conduct monitoring and maintenance after eradication or control measures in order to ensure their effectiveness and to assess whether active restoration measures are needed.

In **Chapter 4** we conducted a global literature review to quantitatively assess the impacts of invaders across a variety of ecosystems around the world and the consequences of their removal on the native plant community. By statistical meta-analysis, we have quantified the magnitude and direction of the change in native plant species richness and abundance with invasion and after alien plant removal. Specifically, we have determined whether (1) alien species cause a decline in the richness and abundance of native plants, (2) the sites where alien plants have been removed are similar to reference sites (not invaded and not managed), and finally (3) we have identified the main methods used when monitoring and evaluating control measures.

In **Chapter 5** we used as a case study the removal of *Carpobrotus sp.*, one of the most widespread and problematic invasive plants along the Spanish coasts. We have evaluated the short-term effectiveness of its removal as well as the recovery of the native vegetation in different ecosystems of coastal Andalucía. In this study, we have compared plots invaded by *Carpobrotus*, plots where *Carpobrotus* was removed manually and reference plots.

Finally, in **Chapter 6** we discuss the main results and conclusions of the thesis.

The four central chapters are presented in scientific paper format, which entails some redundancy in the introductions' information of the different chapters, but the references and appendices have been assembled at the end of the thesis to avoid repetitions and to facilitate readability. The central chapters and the general discussion are written entirely in English. The conclusions of the thesis are presented in Catalan and in English as I have done in this general introduction.

# Chapter 2

# An assessment of stakeholder perceptions and management of noxious alien plants in Spain[2]

[2] Andreu J, Vilà M, Hulme PE (2009) An assessment of stakeholder perceptions and management of noxious alien plants in Spain. *Environmental Mangement* 43: 1244-1255.

## Abstract

Despite biological invasions being a worldwide phenomenon causing significant ecological, economic and human welfare impacts, there is limited understanding regarding how environmental managers perceive the problem and subsequently manage alien species. Spanish environmental managers were surveyed using questionnaires in order to (1) analyze the extent to which they perceive plant invasions as a problem; (2) identify the status, occurrence and impacts of noxious alien plant species; (3) assess current effort and expenditure targeting alien plant management; and finally (4) identify the criteria they use to set priorities for management. In comparison to other environmental concerns, plant invasions are perceived as only moderately problematic and mechanical control is the most valued and frequently used strategy to cope with plant invasions in Spain. Based on 70 questionnaires received, 193 species are considered noxious, 109 of which have been the subject of management activities. More than 90% of species are found in at least one protected area. According to respondents, the most frequently managed species are the most widespread across administrative regions and the ones perceived as causing the highest impacts. The perception of impact seems to be independent of their invasion status, since only half of the species identified as noxious are believed to be invasive in Spain, while 43% of species thought to only be casual aliens are causing a high impact. Records of management costs are poor and the limited data indicate that the total actual expenditure amounts to 50,492,437 €, in the last decade. The majority of respondents stated that management measures are insufficient to control alien plants due to limited economic resources, lack of public awareness and support, and an absence of coordination among different public administrations. Managers also expressed their concern about the fact that much scientific research is concerned with the ecology of alien plants rather than with specific cost-efficient strategies to manage alien species.

## Introduction

Biological invasions are considered to be one of the most serious threats to global biodiversity and ecosystems integrity (Vitousek et al. 1997, Parker et al. 1999, Mack et al. 2000). The introduction of alien species not only generates ecological impacts, but also has economic and human welfare consequences (McNeely 2001). The direct economic costs can be large either due to losses in production of natural resources, damage to infrastructures, or to subsequent costs arising from the management of invasive species. Pimentel et al. (2005) have estimated that economic damage associated with alien species impacts and their control in USA exceeds $120 billion per year.

Economic valuation is a useful tool for policy-makers to guide actions targeting biodiversity conservation priorities and raise public awareness (Costanza et al. 1997, Zavaleta 2000, Brauer 2003, McConnachie et al. 2003, Born et al. 2005, Hulme 2006). However, the economic impacts of alien species are still poorly known for Europe (Hulme 2007), and are often limited to the costs of a single species in a specific location (Vilà et al. 2010). For plants, there are a few published papers on the costs of eradication and control for particular well-known invasive species, such as *Fallopia spp.* (Child et al. 1998); *Rhododendron ponticum* (Dehnen-Schumutz et al. 2004) and *Crassula helmsii* (Shaw 2003) in the UK; *Fallopia spp.* in the Czech Republic (Krívanek 2006), and *Heracleum mantegazzianum* in Denmark (Nielsen et al. 2005). In the UK, Williamson (2002) has calculated the costs of thirty agricultural weeds and invasive plants based on estimated annual expenditure on herbicides. In Germany, Reinhardt et al. (2003) have estimated the total costs of the management of major invasive plants in the country. However, these extrapolations are based on estimated costs of particular species in a certain area. An assessment of the actual costs of invasive plants in natural areas has not been undertaken yet in any European state.

While a quantification of costs may be useful, it must go hand-in-hand with an understanding of the limitations, impediments and opportunities for effective management. Much of the time, managers have to deal with limited resources, which in turn require that choices must be made regarding where best to focus management efforts, and which alien species to prioritize for management (Westman 1990). Therefore, there is a need to understand more fully the implications of the perceptions held by managers regarding biological invasions and how the scientific information percolates through to management decisions (Hulme 2003, Bardsley & Edwards-Jones 2007, Daehler 2008, García-Llorente et al. 2008).

Questionnaire surveys have been used successfully to assess human perceptions of alien species, the risks they pose and the options for control in certain areas (Perrins et al. 1992, Kowarik & Schepker 1998, Williamson 1998, Bardsley & Edwards-Jones 2006, 2007, Daehler 2008, García-Llorente et al. 2008). We adopt this approach with environmental managers in Spain in order to gauge their perception regarding plant invasions and to gather information about management activities. We consider as noxious those alien plants occurring in natural areas and assumed to cause some ecological (i.e., competition with native species, hybridization, changes in ecosystem structure, etc.), economic (i.e., losses on produces, infrastructure damage, management costs, etc.), social (i.e., reduction in aesthetical values, impediments for recreation or navigation, landscape alteration, etc.) or health impacts (i.e., allergies or skin rushes). In particular, we assessed (1) whether senior environmental managers perceive invasions as a problem; (2) the status, occurrence and perceived impacts of noxious alien plants in natural areas; (3) the management activities undertaken in order to prevent or control noxious alien plants and their associated costs; and (4) the criteria managers use to set priorities for management. Regarding this last goal the following questions were addressed: (a) Are those alien species subject to management regarded as invasive? (b) Are managed species the most widely distributed? (c) What type and magnitude of impacts do these alien plants cause? We also discuss whether there is concordance on regional species occurrence between the information provided by environmental managers and the most updated scientific knowledge available on alien plants in Spain (Sanz-Elorza et al. 2004).

## Methods

### *Study region*

Spain is divided into 19 administrative regions: 17 Autonomous Communities (ACs hereafter) and two autonomous cities - Ceuta and Melilla, located in northern Africa. The ACs are subdivided into 50 provinces. Each AC possesses an environmental department, which is the primary environmental administrative body of the region and is responsible for the management of its natural areas. However, specific management activities are usually coordinated by the relevant provincial delegations in each AC.

The establishment and management of protected areas are under the jurisdiction of the environmental departments of each AC. There are 13 national parks, which receive the highest protection status in Spain and 120 natural parks, the second highest protection status. These protected areas are of great ecological, scientific and educational value, encompassing an enormous range of ecosystem types, from arid salty flats and dunes to mountain ranges and woodlands.

The Spanish Environmental Ministry, following the recommendations of the Convention on Biological Diversity (http://www.biodivorg), launched in 1998 the "Spanish Strategy for the conservation and sustainable use of the biodiversity", in which they proposed "the elaboration and harmonization of legal and technical resources needed to control, and avoid the introduction of alien species that threaten biodiversity" (http://www.mma.es/conserv_nat/planes.htm). The Law of Natural Heritage and Biodiversity (42/2007) includes specific requirements for the prevention and control of invasive alien species. The responsibility for such requirements falls to the ACs.

Between 10 and 14% of the total Spanish flora is non-native (Sanz-Elorza & Sobrino 2002, Dana et al. 2003). According to the first national compendium of alien plants in Spain (Sanz-Elorza et al. 2004), a total of 998 alien species have been identified in Spain and, following Pyšek et al. (2004), 123 (12%) are considered invasive, 42% naturalized, 38% casual and the remaining 8% correspond to alien species with unknown status.

### *Questionnaire survey*

Questionnaires were used to assess the perceptions, impacts and management of plant invasions in natural areas in Spain. Respondents were senior managers of all public environmental administrations with responsibility for biodiversity conservation and management of natural areas at both national and AC levels. The environmental sectors assessed included forestry, water management, nature conservation, coastal protection and urban green departments. The agricultural sector was not surveyed, since alien plants in arable fields are not usually managed specifically, but only as components of the total weed flora.

Environmental administrations were first contacted by telephone, in order to identify the person with responsibility for decisions regarding the management of biological invasions. Specifically we contacted the environmental departments of the 19 ACs and their respective provincial delegations in those cases where the information was not centralized. The survey also included the contact with all 13 national parks and 120 natural parks. In order to take into account all potential natural areas where plant invasions could be a problem, we also contacted 7 hydrographic confederations (responsible for catchment management) and 12 coastal protection administrations. In sum, all high-level public administrations with responsibility in conservation were contacted.

The recipients of our first telephone interview and their contact details were identified using information obtained from the Internet, personal contacts or directly calling the environmental administration and asking for the senior official responsible for biodiversity conservation or natural areas management. We also used the "snowballing" method whereby the respondent put us in touch with other secondary public bodies (i.e., county councils and municipalities) with responsibilities relevant to plant invasions (Bardsley & Edwards-Jones 2006).

Subsequently, a structured questionnaire was sent to all senior environmental managers that at least had some responsibility relating to alien plant management. All recipients of the questionnaire were informed about our aim of gauging an institutional, rather than a personal response. The questionnaire comprised two parts: a general section to assess institutional opinions and perceptions of plant invasions in relation to other environmental threats in Spain, and a specific section for any alien plant species they described as being noxious in their areas of responsibility (Appendix I).

From April 2006 to February 2007 questionnaires were sent to 90 institutions with a 78% response rate. This can be considered a high response rate compared to other studies (e.g., 58% response rate in Kowarik & Schepker 1998). Thus, our study can be regarded as representative of current perspectives and activities relating to alien plant management in Spain. We are also confident that we contacted the respondents with the highest level of knowledge regarding alien species in their departments. Each noxious alien plant mentioned by each respondent in the questionnaire's specific section was treated as a separate case. In total, we obtained information on 255 cases and 212 of them contained information on management strategies.

### *Data analysis*

The relative importance of biological invasions compared to other environmental threats (natural habitat loss, habitat fragmentation, wildfire, climate change, pollution, urbanization, land use change) and the perceived effectiveness of four different management strategies against invasions (legislation reinforcement, education and outreach, entry prevention, direct population control) were compared with Kruskal-Wallis tests. We perform a multiple comparison test after Kruskal-Wallis using the package 'pgirmess' and the 'Kruskalmc' procedure under R version 2.6.2

We classified all noxious species identified by managers as being casual, naturalized or invasive in Spain, following Sanz-Elorza et al. (2004), in order to assess whether invasive species were considered more frequently as noxious. A $X^2$ test was used to compare differences between managed and unmanaged noxious species in relation to their invasion status as well as between invasive and casual species in relation to the magnitude of their impact (i.e., high, intermediate, low).

The number of ACs or protected areas in which a species occurred was used as a measure of how widespread the species was in Spain. Differences in the national distribution of managed and unmanaged species were compared using a Mann-Whitney test. Regression analyses were used to contrast the number of ACs where a noxious species had been recorded with its known distribution in Spain (Sanz-Elorza et al. 2004). To assess whether managed species were among the most widely distributed alien plants across Spain, linear regression analysis was performed between the number of ACs or protected areas where a noxious alien plant was present and the number where it was actually managed. In order to know whether management was directed towards species causing a particular ecological impact (ecological, economic, social, human health), differences in the proportion of unmanaged and managed species causing different ecological impacts were compared using a chi-square test.

Finally, we tested the consistency among responses to the type of impacts (i.e., ecological, social, economic, health) and management approach applied (i.e., prevention, eradication, containment and restoration). For impacts, we selected the 10 most widely distributed noxious species across the ACs and compared the similarity of responses by respondents using the Sorensen Similarity Index between all possible paired comparisons. For example, the Sorensen Similarity between respondent A and B, was calculated as S = (2 x

C) / (2C + A + B); where C is the number of responses common to both respondents; A is the number of responses mentioned by only respondent A, and B is the number of responses only mentioned by respondent B. A similar analysis was undertaken to compare management approaches but in this case the ten most frequently managed species were selected as the basis for comparisons.

All analyses were carried out with the software package STATISTICA 6.0 (StatSoft 2001). Mean values ± standard error are given.

## Results

### *General perceptions regarding the threat of biological invasions*

Significant differences were found in the importance given by respondents to different environmental problems (Kruskal-Wallis, $H = 66.04$; $df = 7$; $P < 0.0001$). While 50% of respondents felt biological invasions were at least a medium priority for management, and over a third stated that this threat was a high priority, on average, biological invasions were perceived as an intermediate threat to biodiversity. Managers manifested greater concern to landscape changes such as habitat loss, urbanization, habitat fragmentation and land use changes, than to wildfires, pollution and climate change. Nonetheless, concern on biological invasions ranked similar to all these environmental problems (Figure 2.1).

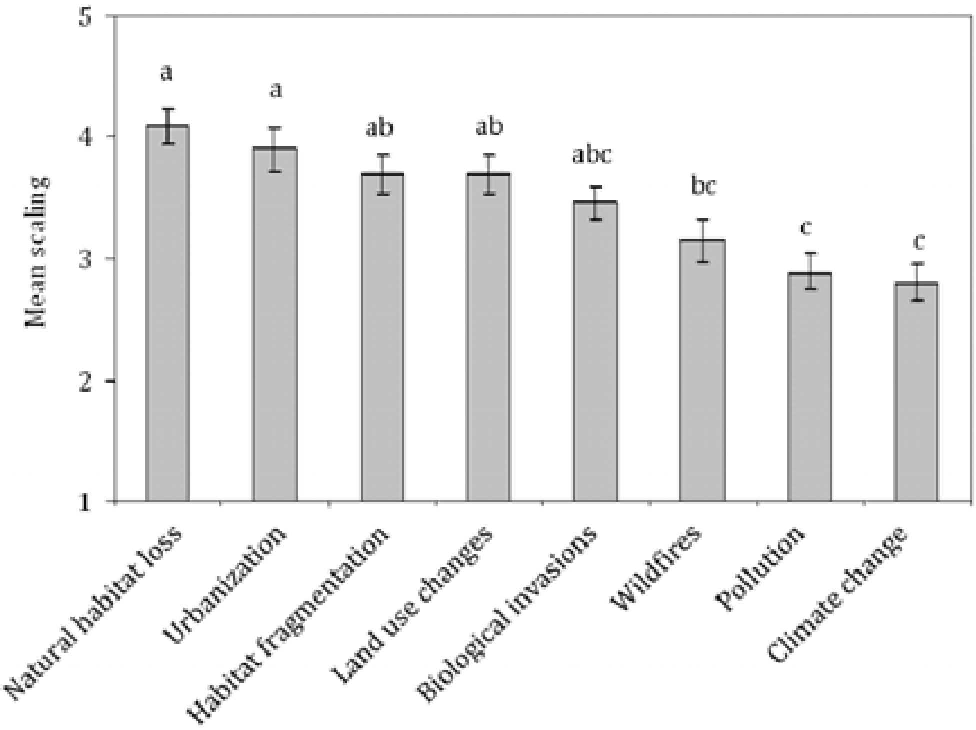



**Figure 2.1** Respondents' perception (mean ± SE) of the importance of the different environmental problems in Spain, on a 1 (not relevant) to 5 (extremely important) point scale. Different letters indicate significant differences among environmental problems.

## *Identity, status and occurrence of noxious alien species*

In total, 193 alien plants were identified as noxious (i.e., listed by respondents as being of concern), yet only a little more than half of these species (109) were the subject of any management (Appendix II). Not all noxious species were alien, 33 (17%) are native species of the Iberian Peninsula (i.e., continental Portugal and Spain), but aliens in Canary Islands. Only 14 of these species were managed in Canary Islands. Most (~50%) species identified as noxious were classed as invasive (Sanz-Elorza et al. 2004), but a significant proportion (~20%) were only classed as casual (Figure 2.2). Overall, of the 123 alien species classified as invasive in Spain, only 60% were identified as noxious by respondents.

There was a significant difference in species status between managed and unmanaged noxious species ($X^2 = 39.11$; df = 3; $P < 0.0001$). However, differences could most probably be explained by a higher proportion of unmanaged species with unknown status (Figure 2.2). In almost half the cases (49.7%) invasive species were not managed despite other naturalized or casual alien species being targeted.

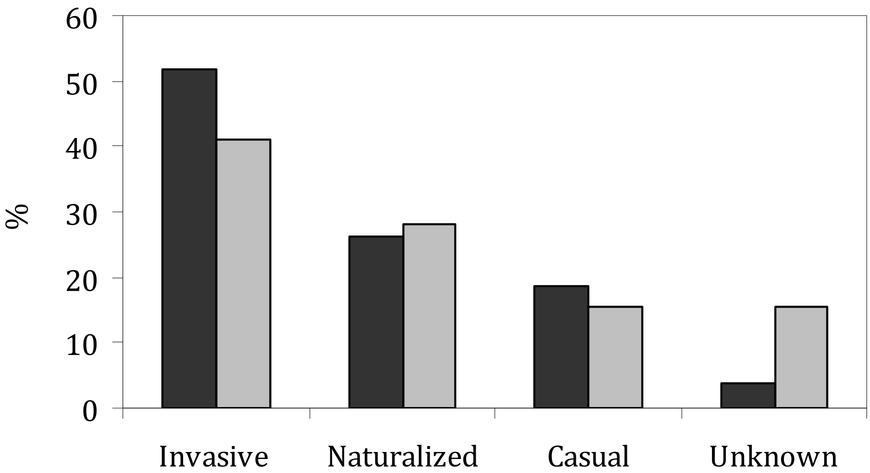



**Figure 2.2** Invasion status classification of managed (black) and unmanaged (grey) species that have been mentioned by respondents according to the Atlas of alien plants of Spain (Sanz-Elorza et al. 2004).

The taxa most frequently identified as noxious were *Carpobrotus spp., Eucalyptus spp., Ailanthus altissima* and *Robinia pseudoacacia* (11 ACs out of 19) followed by *Acacia spp.* (9 ACs) and *Cortaderia selloana* (8 ACs) (Appendix II). This group also included those species most frequently reported as managed: *Carpobrotus spp.* and *Eucalyptus spp*. (8 ACs), *Acacia spp.* (7 ACs) and *Cortaderia selloana* (6 ACs). However, most species, 130 out of 193, were mentioned in only one AC. Managed species were present in more ACs across the country than unmanaged species (Mann-Whitney, Z = 3.15; $P$ = 0.001). In general, although the most widespread species were also the most managed (y = 0.54x-0.12; $R^2$ = 0.65; $P$ < 0.0001), across all species, management only occurred in approximately half of the ACs (Figure 2.3).

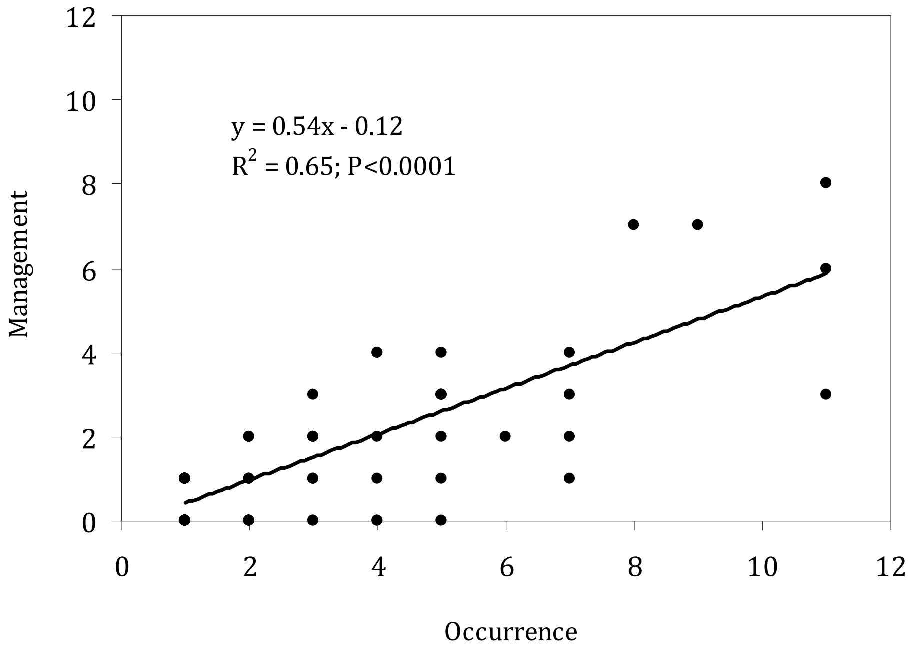



**Figure 2.3** Relationship between occurrence (i.e., number of Autonomous Communities [ACs] of Spain where a particular noxious alien plant is present) and the number of ACs where this species is managed.

Most of the noxious alien species mentioned by respondents (92%) were neophytes (i.e., alien plants introduced after 1500; Pyšek et al. 2004). Nine of them (8%) were archaeophytes (i.e., introduced before 1500; Pyšek et al. 2004). *Ricinus communis, Morus spp.* and *Prunus cerasifera* were managed in some locations, and the magnitude of their impact was high. *Arundo donax* was the only archaeophyte managed in more than one case (4 cases) and in all of them it was reported to have a high impact, mostly on riparian areas.

Considering protected areas, 94% of noxious species were found in at least one protected area. Management was undertaken in a significantly smaller proportion of natural (34 out of 120) than national parks (8 out of 13, $X^2$ = 5.99; df = 1; $P$ < 0.05). In contrast to the pattern for ACs, managed species were no more widespread in protected areas than unmanaged species (Mann-Whitney, Z = 1.52; $P$ = 0.13). Similarly to ACs, while the most widespread species were also the most managed (y=0.56x+0.10; $R^2$ = 0.73; $P$ < 0.0001) no matter how widespread the species, management occurred in approximately half the recorded parks. *Carpobrotus spp.* are also the most widely distributed and most managed taxa in national and natural parks, followed by *Eucalyptus spp.*

Comparison between the occurrence records provided by respondents and the known distribution of species across all ACs in Spain (Sanz-Elorza et al. 2004) reveals that either alien species are under-recorded by respondents, or that species are viewed as noxious by respondents in only a few of the regions where they are found (i.e., *Amaranthus spp., Datura stramonium* or *Xanthium spinosum*; Figure 2.4). However, for some species, such as *Carpobrotus spp., Ailanthus altissima, Pennisetum setaceum, Egeria densa* and *Ludwigia sp.* occurrence records provided by respondents matched the known distributions of these species in Spain.

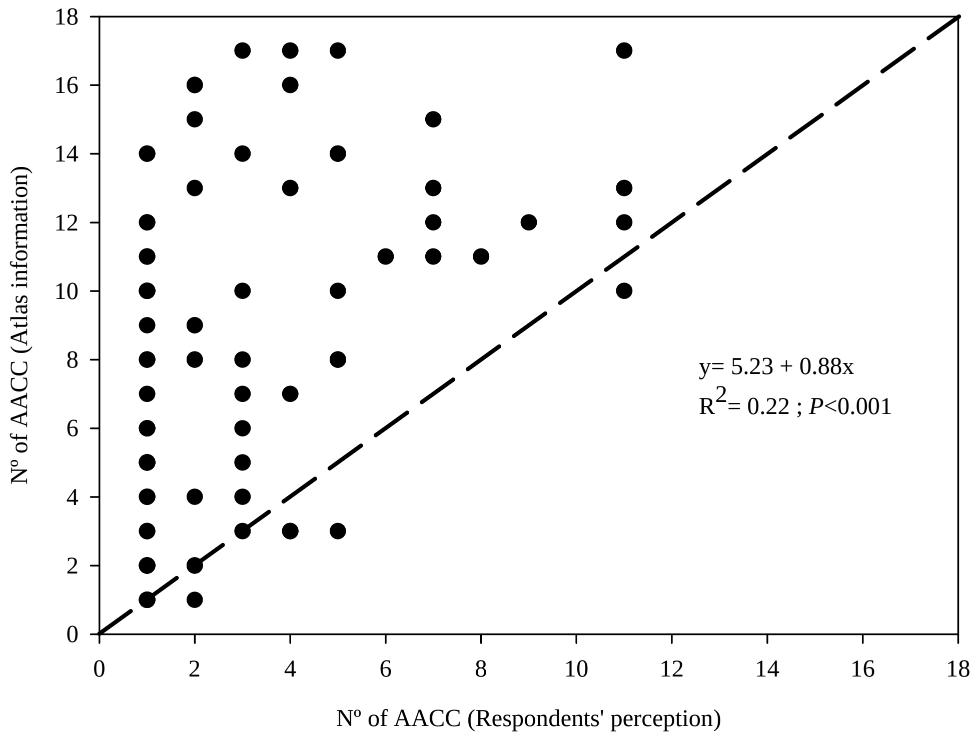



**Figure 2.4** Relationship between respondents' perception about the occurrence of a particular noxious alien plant in terms of number of Autonomous Communities (ACs) where the species have been mentioned and the number of ACs where the species is known to occur (Sanz-Elorza et al. 2004). The line of unity is indicated and represents equivalence between the two sources of information.

## *Perception of impacts of alien species*

Regarding the magnitude of the impacts caused by noxious alien species, 35% of the cases were perceived as causing a high impact on natural areas and 28.5% a low impact. In only 36% of the cases invasive species were perceived to cause a high impact and in 25% of the cases a low impact (Figure 2.5); while in naturalized and casual species cases, 30% and 43% were causing a high impact, respectively (Figure 2.5). No significant differences were found between invasive and casual species in their perceived magnitude of impact ($X^2$ = 4.75; df = 2; $P$ = 0.092).

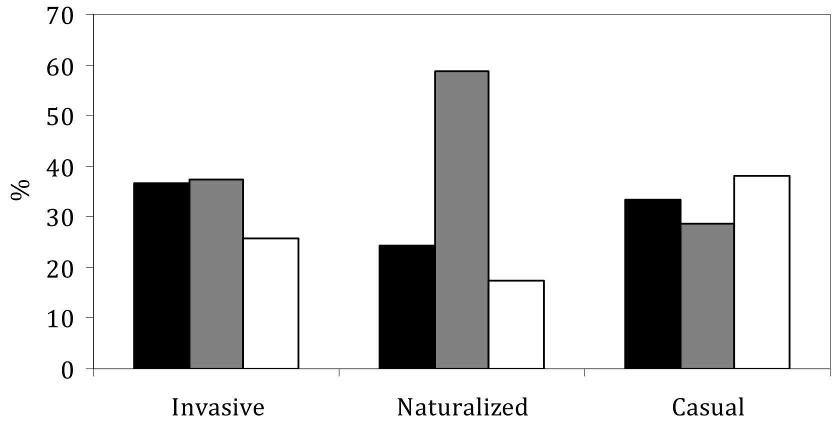



**Figure 2.5** Magnitude of the impact (high: black; intermediate: grey and low: white) perceived by respondents across invasive, naturalized and casual species.

Most cases with a high impact (88%) were being managed, thus the magnitude of the impact could be regarded as a criterion for managers to prioritize the allocation of resources. The remaining 12% were species that despite having a high impact on natural areas were not being managed because control was neither feasible (because they were too widely distributed) or affordable (due to a lack of sufficient funds). Surprisingly, 78% of species with low impacts were managed. In many cases (98%) these species were managed as part of a wider targeting of high impacting species.

All noxious alien plants were causing some ecological impacts. Besides these impacts respondents also mentioned social (23%), economic (9%) and human health (3%) impacts. Respondents appeared to be relatively consistent in their assessment of perceived impacts (Sorensen Similarity Index, 0.74 ± 0.04).

The main ecological impacts mentioned were competition with native species for space and soil resources, species loss, and changes in the integrity and stability of ecosystems. Other impacts included indirect effects upon the fauna by changing their behavior or modifying the habitat, changes in the composition and structure of riparian forests, soil erosion and degradation, increment of biomass and flammability and finally, water quality deterioration and eutrophication. No significant differences were found between unmanaged and managed species in the ecological impacts they cause ($X^2 = 0.322$; df = 8; $P = 0.99$; Figure 2.6). Therefore, the type of ecological impact does not seem to be a criterion for management priorities.

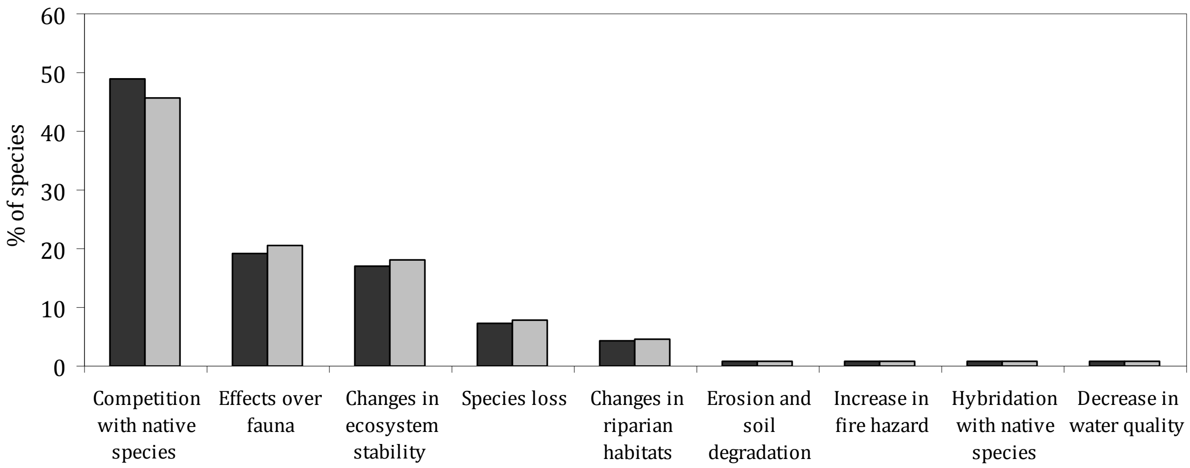



**Figure 2.6** Respondents' perception of the ecological impacts induced by noxious (black) and managed (grey) alien plants in Spain.

Respondents were also asked to name native species that have been negatively affected by aliens. According to respondents, *Carpobrotus spp.* in the natural park of Cap de Creus (Catalonia) outcompetes with *Limonium gerondense, Armeria ruscinonensis, Astragalus massiliensis* and *Seseli farrenyi*, causing in some areas its local displacement. In Isla Grossa (Murcia), *Carpobrotus spp., Acacia spp*. and *Agave americana* are thought to compete with *Lycium intricatum*, *Salsola spp.* and *Withania frutescens*. The presence of *Azolla spp.* in the Miño River, leads to a loss in the cover of *Magnopotamion* and *Parvopotamion* vegetation types by occupying the same ecological niche. *Pennisetum setaceum* in Fuerteventura Island (Canary Islands) outcompetes *Launaea arborescens, Euphorbia balsamifera, Euphorbia regis jubae, Suaeda sp., Salsola spp.* in shrublands and "cardonal-tabaidal" habitats.

### *Management and costs of alien species*

The five management activities were prioritized in the following order: direct control, prevention, education and outreach with legislation being perceived as the least relevant and efficient (Figure 2.7). The main goal of management activities appeared to be containment or population control (41%), or the complete eradication of the species (37%). Prevention through legislation, education or communication with the general public has been less frequently used in Spain (22%). There was considerable variation among respondents in the management strategies applied to a particular species (Sorensen Similarity Index = 0.54 ± 0.03). The primary constraints to alien species management were: limited economic resources (28%), insufficient coordination among administrations (22%), lack of public awareness (16%), negligible legislation (14%), paucity of research on efficient management strategies (9%), absence of long-term monitoring (6%) and few guidelines for priorization (4%).

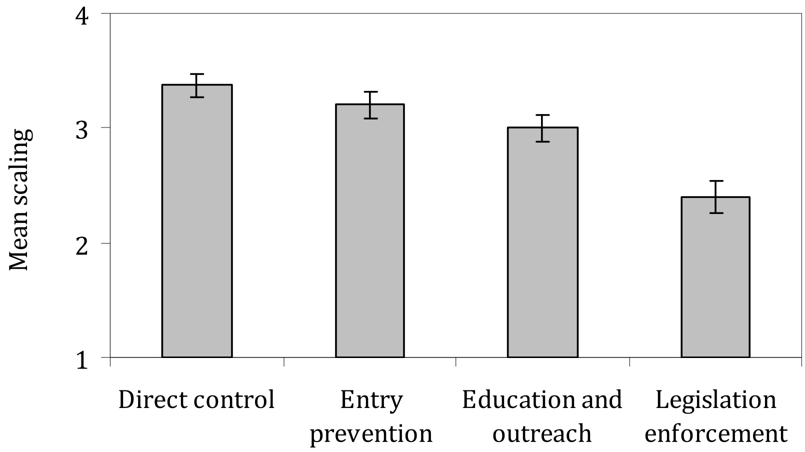



**Figure 2.7** Respondents' perception of the importance of different management strategies for alien plants in Spain, on a 1 (not relevant) to 4 (extremely important) point scale.

In most cases mechanical methods (71%) have been used in management since they are considered the least harmful to the environment, but in 25% of the cases mechanical methods combined with herbicides (usually glyphosate) were used. Only 3% of the cases applied solely chemical methods. In Spain, there has been no attempt to use biological control agents for managing noxious alien plants. Under half of all management activities

(42%) were carried out annually, with 41% through external contracts and 15% via volunteers.

In the great majority of cases (85%) control measures were followed by an annual monitoring in order to detect reinfestation. However, only in very few cases the monitoring was undertaken with long-term goals in mind or using standardized indicators. Restoration of habitats previously invaded, has not been frequently undertaken in Spain (29%). Restoration efforts have generally followed management of *Carpobrotus spp., Eucalyptus spp., Agave spp., Ageratina spp., Ailanthus atissima* and *Acacia spp.*

In general, management activities achieved a significant reduction in the distribution of the alien species. Almost half of the cases, where management has been applied (46%), were effective in reducing the area of distribution, although in only 13% of the cases was the alien species totally eradicated. Rarely has control been completely ineffective (3%), but on six occasions the species have continued to increase despite the control measures applied.

In terms of monetary costs, estimates could only be obtained for the direct expenditure on management activities rather than indirect costs on ecosystem services. Only 41% of management cases provided estimates of costs, but these were largely gross costs relating to control, rather than prevention and restoration. Total expenditure on management amounted to 50,492,437 €, although annual costs could not be estimated the total expenditure probably occurred over the last decade. Over 95% of the expenditure is targeted at five species (Table 2.1). Prevention costs were specified in only 7 cases and amounted to less than 1% of total costs (381,744 €), while expenditure on restoration accounted to around 2% (1,088,310 €) of total management expenditure.

**Table 2.1** Overall management-costs of the alien species mentioned by the respondents.

| **Invasive species** | **Management costs** |
|---|---:|
| *Eucalyptus spp.* | 31.528.594 € |
| *Eichhornia crassipes* | 6.700.000 € |
| *Pennisetum setaceum* | 6.203.300 € |
| *Carpobrotus spp.* | 2.886.683 € |
| *Azolla filiculoides* | 1.000.000 € |
| *Acacia spp.* | 90.000 € |
| *Rumex lunaria* | 86.000 € |
| *Agave spp.* | 57.000 € |
| *Ailanthus altissima* | 28.675 € |
| *Ageratina adenophora* | 23.109 € |
| *Senecio inaequidens* | 19.600 € |
| *Arctotheca calendula* | 15.000 € |
| *Cortaderia selloana* | 8.600 € |
| *Plectranthus australis* | 6.251 € |
| *Fallopia aubertii* | 6.000 € |
| *Pittosporum tobira* | 6.000 € |
| *Opuntia spp.* | 4.000 € |
| *Hakea sericea* | 2.000 € |
| *Ambrosia spp.* | 1.000 € |
| *Panicum repens* | 1.000 € |
| *Myoporum spp.* | 400 € |
| *Lonicera japonica* | 200 € |
| **Several species** | 1.819.025 € |
| **TOTAL** | 50.492.437 € |

## Discussion

Environmental managers in Spain are clearly aware of the risks posed by biological invasions, which ranked similarly to any other environmental problems. This awareness is encouraging given that biological invasions have not been perceived as a problem by Spanish public authorities until very recently, or included in the environmental agenda (Martín 2001, Capdevila-Argüelles et al. 2006). This is not surprising as even in regions such as Hawaii, where the threat of biological invasions is of global significance, public concern can be rather limited (Daehler 2008). However, Mediterranean ecosystems may be particularly resistant to plant invasions, even by those species regarded as major threats to biodiversity (Vilà et al. 2008). As a consequence, the institutional response in Spain was still primarily to undertake responsive actions targeting mechanical or chemical control at a local scale rather than addressing broader legislative issues that might tackle prevention (Hulme 2006, Lodge et al. 2006, Smith et al. 2006).

Environmental managers clearly set priorities for management within their area of responsibility but such decisions were not based on the status of the species at a national scale. Rather, decisions were made based on local perceptions of abundance, distribution, and perceived impact. Invasion status was not related to the magnitude of the impact caused by a certain species and is consistent with the finding that species capable of rapid colonization are, in general, no more likely to have impacts on biodiversity (Ricciardi & Cohen 2007). This has important implications regarding national coordination of invasive species management: a species deemed of concern at a national scale simply because they are widespread may not be a priority at the local scale of a national park and diverting resources to manage such a species may not always be sensible. One solution would be to establish national priorities based on an integrative risk index that combines information on both local and regional abundance (Hulme et al. 2008a).

Management did tend to target species perceived as noxious by the majority of respondents (i.e., higher occurrence). However, there are exceptions, such as *Ailanthus altissima*, *Robinia pseudoacacia*, *Arundo donax, Opuntia spp.,* or *Oxalis pes-caprae* that despite being frequently identified as noxious are rarely managed. Many of these species represent some of the most problematic species in the Mediterranean due to the difficulty and cost of control (Hulme et al. 2008a). Environmental managers were consistent in their perceptions of impacts of alien plants, most supporting the idea that these species outcompeted native species, though the specific type of impact did not influence management activities. There is evidence in the Mediterranean than the niches of alien and native plants do overlap (Lambdon et al. 2008a), and thus competitive interactions may be important and lead to biotic homogenization (Lambdon et al. 2008b).

In contrast to the perception of impacts, environmental managers differed in their views regarding how best to manage invasions and this probably reflects a lack of guidance and limited resources that perhaps results in less effective management (Westman 1990, Bardsley & Edward-Jones 2006). Control programs often have short-term goals and few supervise the longer term success of actions or use standardized indicators to monitor management success (e.g., native vegetation regeneration). Hulme (2003) emphasizes that the application of ecological knowledge to the management of biological invasions can lead to the most cost-effective strategies. Unfortunately, "management" is not a prevalent keyword topic in research on biological invasions (Pyšek et al. 2006). Moreover, simply eliminating the alien plant from an ecosystem may not always lead to restoration of the original community and sites can often be colonized by other alien species (Simberloff 2003, Hulme & Bremner 2006). An ecosystem perspective of invasion management that addressees both the drivers of invasions as well as the target species control is required (Hulme 2006).

The main bottlenecks encountered when trying to obtain economic costs were as follows: (1) there was a lack of recorded expenditures, (2) the management of aliens was only one of many management tasks and thus not distinguished in budgets and (3) where targeted management occurred, it was not specific to a single species and location. As a result, the estimate 50,492,437 € is unreliable and probably underestimates the true direct cost. This figure does not account for indirect costs to forest or pasture production, landscape changes, damage to infrastructure or recreational opportunities. If monetary values could be assigned to losses in biodiversity, ecosystem services, and aesthetics, these costs would undoubtedly be several times higher than reported (Zavaleta 2000, Pimentel et al. 2005, Binimelis et al. 2007).

## Conclusions

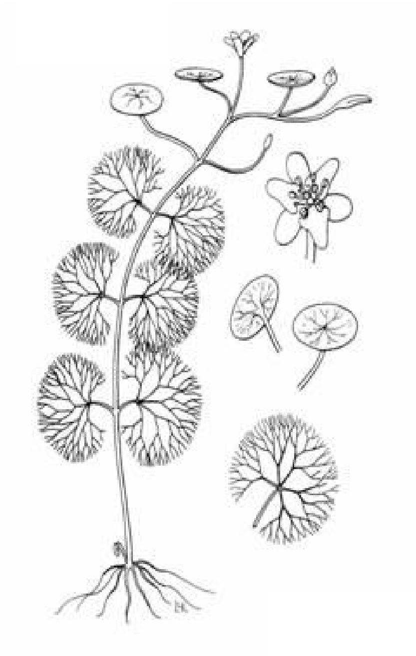
Human perception plays a strong role in addressing the issue of biological invasions (de Poorter 2001, Larson 2007, Daehler 2008). We have presented the first national assessment of these perceptions in Europe. Biological invasions are considered by Spanish environmental managers a medium priority problem and a total of 109 noxious alien species are being managed due to its impacts on natural areas. However, there remains a pressing need to raise awareness about the impacts of alien species among the general public, environmental managers and policy-makers (Daehler 2008). Collaboration between academic research and environmental managers is required in order to achieve an efficient management of alien plants and to protect environmental integrity and native species diversity. To date, attempts to use science to advise and improve the cost-effectiveness of management strategies are few (Moody & Mack 1988, Wadsworth et al. 2000, Hulme 2003, Taylor & Hastings 2004). If the putative costs identified in this study are representative of the general situation, then there is substantial benefit to be gained by investing in better management strategies.

## Acknowledgments

We thank all participants of the inquiry (http:// www.creaf.uab.es/propies/jara/Apéndice artículo Ecosistemas.pdf), too many to name here, and three anonymous reviewers for helpful comments on a previous version of the manuscript. This study has been partially funded by the projects ALARM (Assessing large-scale environmental risks for biodiversity with tested methods - GOCE-CT-2003-506675; http://www.alarmproject.net; Settele et al. 2005) and DAISIE (Delivering alien invasive species inventories for Europe SSPI-CT-2003-511202; http://www.europe-aliens.org) of the 6th Framework Programme of the European Commission, RINVE (Determinantes biológicos del riesgo de invasiones vegetales) from the Spanish Ministerio de Educación y Ciencia and the CONSOLIDER-INGENIO 2010 project: Spanish woodlands and global change: threats and opportunities (MONTES-CSD2008-00040).

# Chapter 3

# Risk analysis of potential invasive plants in Spain[3]



[3] Andreu J, Vilà M (2010) Risk analysis of potential invasive plants in Spain. *Journal for Nature Conservation* 18: 34–44.

## Abstract

Once alien species become established in a new region, they are extremely difficult to eradicate or control, suggesting an urgent need for the development of early warning systems to determine the probability of a given species becoming invasive. Risk assessment schemes are valuable tools to diminish the risk of invasion and to concentrate resources on preventing the entrance and spread of those species with higher risk of invasion. For many European countries, plant species not yet introduced to the country and with high invasive potential have not been identified. The present study aims to identify and rank alien plant species that could potentially become invasive in Spain if introduced. As a first step, a plant data set was pre-selected for screening, containing invasive plants in neighbouring countries and in other Mediterranean regions of the world, not present in Spain. A preliminary list of 80 species was obtained, being Leguminosae the most represented family (15%) and gardening (62%) the most common pathway of introduction. As for the potential European Nature Information System (EUNIS) habitats to invade, heathland, scrub and tundra habitats types (F; 19%), followed by constructed, industrial and artificial habitats (J; 14%) and woodland and forest habitats (G; 13%) seem to be the habitats most at risk despite that F and G habitats are currently the ones least invaded in Spain. We ranked these potential invasive species using the Australian Weed Risk Assessment system and a Risk Assessment for Central Europe developed by Weber & Gut (2004). Most species received intermediate values in both tests. The species with higher scores, were mainly aquatic plants, and should be prohibited or kept out of trade. *Chromolaena odorata* (Asteraceae) obtained the highest score in both tests and therefore is the species with the highest risk to become invasive in Spain if introduced.

## Introduction

Invasion by alien species represents one of the greatest threats to biodiversity worldwide and is considered a major component of global change (Mack et al. 2000, Mooney & Hobbs 2000). In addition to affecting ecosystems and contributing to the local extinction of native species, invasive alien species can also cause major socio-economic damage (Pimentel et al. 2005). The introduction of alien species has increased dramatically in frequency and extent in recent decades, as human movements have become more global and international trade has increased (McNeely et al. 2001). This trend has entailed an increase in the likelihood of new invasion events with subsequent negative ecological and socioeconomic impacts (Vitousek et al. 1997, Mack et al. 2000, Levine et al. 2003, Pimentel et al. 2005).

Once introduced plant species are established in a new region, they are extremely difficult to eradicate or control (Duncan et al. 2003, Rejmánek et al. 2005a). Thus, preventing new alien invasions is, by far, the most environmentally desirable and cost-effective management method (Wittenberg & Cock 2001). Consequently, there is an urgent need for the development of early warning systems to determine the probability of a given species becoming invasive (Panetta & Scanlan 1995, Groves et al. 2001).

For plants, research on (1) the historical events related to introduction (Pyšek & Jarosík 2005, Rejmánek et al. 2005a, Pyšek & Richardson 2007), (2) the key species traits associated with invasive species (Rejmanek & Richardson 1996, Reichard & Hamilton 1997, Goodwin et al. 1999) and (3) the characteristics of invaded habitats (Burke & Grime 1996, Lonsdale 1999) have provided the basic information to predict the invasion success in the new region (Williamson 1999, Richardson & Pyšek 2006, Pyšek & Richardson 2007), and therefore for risk assessment analysis (Daehler & Carino 2000, Wittenberg & Cock 2001, Keller et al. 2007).

Risk assessment schemes are science-based predictions which attempt to identify species that have not yet been introduced to a region but have a high likelihood of becoming invasive (Whitney & Gabler 2008). The implementation of risk assessment protocols produces net economic benefits (Keller et al. 2007), but only few countries, such as Australia and New Zealand, have implemented science-based risk assessment schemes as a screening routine to detect potential invasive species posing environmental and economic hazards.

In Spain, potential invasive plants not yet introduced in the country have not been identified, despite that the Law of Natural Heritage and Biodiversity (42/2007), issued by the Spanish Environmental Ministry (http://www.boe.es/aeboe/), mentions the need for prevention and management of those invasive species which threaten native species, natural habitats and economic resources. The present study aims to identify and rank alien plant species that, if introduced, could potentially become invasive in Spain. This research is a basic tool to reinforce gardening, landscaping and plant trade regulations. We also provide information on the basic traits of these species in terms of taxonomy, origin, life-form, and potential habitats to invade compared to invasive species that have already become established in the country (Sanz-Elorza et al. 2004).

Species ranking has been performed by applying two different risk assessment protocols: the Australian Weed Risk Assessment system (hereafter 'WRA', Pheloung et al. 1999) and a Risk Assessment for Central Europe developed by Weber & Gut (2004) (hereafter, 'WG-WRA'). The first one has been selected due to its success and consistency in different regions (Gordon et al. 2008). The second protocol has been chosen because it has been developed specifically for Europe, albeit in central Europe, and uses a similar quantitative grading system than WRA. Both protocols rate species with an index as a measure of invasive potential, facilitating the comparison between them. We compared whether results were consistent among the two tests and discussed main differences.

## Methods

### *Preselection of species*

A plant data set was pre-selected for screening. This plant data set comprised all invasive plants of neighboring countries and Mediterranean regions, but not yet present in Spain. All plant species listed as invasive in Portugal, France, Italy and in the Mediterranean Basin areas of Northern African countries, as well as invasive species in other Mediterranean regions of the world (i.e., Chile, California, Australia and South Africa) were included in the list.

Although, Spain houses a heterogeneous climatic mosaic being oceanic in the North, alpine at high altitudes and somehow continental in the central plateau (but far less extreme than the Central Europe continental climate) most of its territory is influenced by

Mediterranean climate (Ninyerola et al. 2000). Moreover, hot spots of invasive plant richness in Spain are localized in coastal areas with Mediterranean climate (Pino et al. 2005, Gassó et al. 2009a) and very few are located in the central plateau. As Spanish invasive flora is mainly dominated by species of tropical and subtropical origin, most of them are presumably unable to complete their life-cycle in winter cold areas with continental climate (Casasayas 1990, Sanz-Elorza et al. 2004). These are the main reasons for choosing the Mediterranean climate as the most representative one of Spain in relation to invasive plant richness.

In order to compile the plant data set of potential invasive plants for Spain, online databases and scientific papers were consulted (Vilà et al. 1999, Teillier et al. 2003, Figueroa et al. 2004). The list was enlarged by plant species not present in natural areas of Spain listed in the IUCN's list of the 100 World's Worst Invasive Alien Species (http://www.issg.org/database), the DAISIE 100 of the worst invasive species of Europe (http://www.europe-aliens.org/) and the European and Mediterranean Plant Protection Organization (EPPO) quarantine lists (http://www.eppo.org/).

Subsequently, all species considered invasive in more than one country/region were selected for screening. A preliminary list of 80 species was obtained. For each species, the life form, the region of origin and the pathways of introduction were specified. Moreover, first European Nature Information System (EUNIS) category (http://eunis.eea.europa.eu/) habitat types where occurring, both in their native and introduced range, were gathered in order to identify which habitat types were more likely to be invaded. The EUNIS habitat classification system has been used in this study because it covers all types of natural and artificial habitats in Europe and it is widely used by the European scientifics and conservationists to standardize habitat types. Finally, the type of impact (i.e., ecological, socioeconomic or human health impact) and the economic sector affected by the species: conservation, agriculture, cattle rising, recreation, fishing, navigation, health, irrigation and drainage was determined through literature review.

### *Risk assessment schemes description*

The WRA is one of the best known protocols to date (Pheloung et al. 1999). It is a computer based plant screening method producing a score for weediness or invasive potential, which can be converted to an entry recommendation for a specified taxon. It has been developed for and currently is in regulatory implementation in Australia and New Zealand. It consists of 49 questions divided into sections on biogeography, biology/ecology, and traits contributing to invasiveness (Pheloung 1995). Depending on the answer, each question is awarded between –3 and 5 points (most –1 to 1), and the final WRA score is the sum of points for all answered questions. This final score, ranging potentially from –14 (benign taxa) to 29 (maximum risk), leads to one of three outcomes: the species is accepted for introduction (< 1 total points); rejected (> 6 points); or recommended for further evaluation of invasive potential (1–6 points). A minimum of 10 answers are needed for a species to be evaluated: at least two in the biogeography section, two in traits section and six in biology/ecology. This risk assessment protocol has been chosen because it exhibits the highest accuracy when compared to others (Gordon et al. 2008); it has been validated against a large number of already introduced species and effectively discriminates invasive from non-invasive species (Gordon et al. 2008). This validation has been applied with success for Hawaii (Daehler & Carino 2000), the Pacific Islands (Daehler et al. 2004), the Czech Republic (Krivánek & Pyšek 2006), the Bonin (Ogasawara) Islands of Japan (Kato et al. 2006), the state of Florida, USA (Gordon et al. 2008) and Spain (Gassó et al. 2009b). In our study, questions related to geography and climate, were modified to reflect the conditions of the target area. Suitability of species to Australian climate was changed to suitability to Mediterranean climate (question 2.01; see Pheloung et al. 1999). Question 5.03 (Pheloung et al. 1999) has also been modified: "Nitrogen fixing woody plants" to "Nitrogen fixing plants" as these are an important component of Spanish alien flora, many such species being very abundant in ruderal, disturbed habitats (Sanz-Elorza et al. 2004). The test has been answered following instructions presented in Daehler et al. (2004).

For Europe, a few risk assessment systems have been developed. One of the best known is the European Plant Protection Organization (EPPO) Pest Risk Assessment Scheme that covers any pest organisms including plants and classifying them into pest recommended for regulation and plant quarantine, and pests possibly presenting a risk (http://www.eppo.org). This screening procedure primarily developed for plant and insect pests of agricultural habitats is based on expert judgment and does not rank species. Weber & Gut (2004) proposed a rating system to assess the invasion potential of alien plant species according to the specific needs for central Europe. This risk assessment (WG-WRA) consists of a rating system that allocates scores to the species for biogeographical, ecological, and experience-linked aspects. The scores of 12 questions are totaled, and species are classified into ''high risk'' (28-39 points), ''intermediate risk'' (21-27 points), and ''low risk'' (3-20 points). This test has been validated with an overall accuracy of 65%.

In this study we used the WRA and the WG-WRA. Questions were answered with information gathered from scientific literature, Internet searches, floras, horticultural manuals and books. Climatic match was decided by determining the origin of the species and its distribution; following criteria developed by Daehler et al. (2004). The climatic match was considered high if the species was native to Mediterranean regions. It was intermediate if the species was not native to Mediterranean regions but was growing successfully in these regions and was considered low if the species was not native to Mediterranean regions and was not growing successfully in these regions.

Taxonomical information and geographical distribution data for Europe and the world were obtained from Flora Europaea online (http://www.rbge.org.uk/forms/fe.html), the online database from the Germplasm Resources Information Network (GRIN; http://www.ars-grin.gov/), the online database from the European Project DAISIE (http://www.europe-aliens.org/) and the information published by EPPO (http://www.eppo.org/). Ecological data were extracted from BioFlor (http://www.ufz.de/biolflor), Plants for a Future (2002) (http://www.comp.leeds.ac.uk/pfaf), species accounts from *Plantas Invasoras em Portugal* (http://www.uc.pt/invasoras), USDA Plants database (http://plants.usda.gov), International Survey of Herbicide Resistant Weeds (http://www.weedscience.org), Global Compendium of Weeds (http://www.hear.org/gcw), Global Invasive Species Database (http://www.issg.org/database/welcome), Weeds in Australia (http://www.weeds.gov.au), and Ecological Traits of New Zealand Flora (http://ecotraits.landcareresearch.co.nz).

## Results and discussion

### *Characteristics of potential invaders*

From the 80 potentially invasive species selected for further examination, the families with the highest number of species were Leguminosae (15%), followed by Asteraceae (10%) and Poaceae (6.3%). According to the database of the Atlas of invasive alien plants of Spain (Sanz-Elorza et al. 2004) updated with expertise information (Gassó et al. 2009b) the most represented families of invasive species in Spain also belong to these families but in significantly different proportions ($\chi^2_9 = 45.2$; $P < 0.0001$): Asteraceae (18%), Poaceae (14%) and Leguminosae and Cactaceae (9%).

Regarding the life form, most of the screened plants are woody species (47%), followed by perennial herbs (24%), vines (11%), aquatic plants (10%) and annual herbs (8%). These results differ significantly from the growth habits of already existing invasive plants of Spain ($\chi^2_4$ =95.7; $P$ < 0.0001); being perennial herbs (44%) followed by annual herbs (29%), woody species (21%), vines (5%) and, finally, aquatic plants (2%) (Sanz-Elorza et al. 2004). In central Europe, woody plants have been reported as successful invaders compared to other plant groups (Krivánek & Pyšek 2006). Thus, special care must be taken with this group of potential invasive species.

Twenty-nine percent of the potential invaders have originated in Central and/or South America, 28% in Asia, 21% come from North America, 11% from Africa and 8 % from Australia (Figure 3.1). These proportions are significantly different from the ones for invasive species already present in Spain ($\chi^2_4$ = 22.12, P<0.001; Sanz-Elorza et al. 2004). However, America is also the main origin of invasive species already present in Spain (58%), followed by Africa (16%), Asia (15%), Australia (6%) and Europe (3.5%).

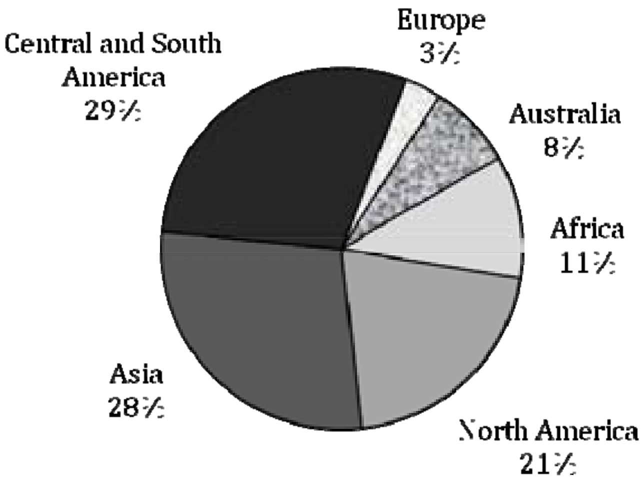



**Figure 3.1** Origin of the 80 potentially invasive species for Spain.

Sixty-two percent of the screened species are ornamental and their main introduction pathway is through gardening. Only 12% and 11% are introduced through agriculture or unintentionally, respectively. The rest are used in acuaculture (6.5 %), silviculture (4.6 %) and restoration (3.7 %). These proportions are significantly different to the ones obtained for invasive plants of Spain, for which the most common pathway of introduction is also gardening (50%) but it is followed by unintentional introduction (27%), agriculture (21%) and silviculture (3%) (Sanz-Elorza et al. 2004; $\chi^2_3$ = 17.55, P<0.001). The fact that most potential invaders could be introduced deliberately rather than accidentally could reduce the risk of invasion given adequate resources and political will to ban their entrance (Hulme et al. 2008b).

As for the potential EUNIS habitats to invade, heathland, scrub and tundra habitats (F; 19%), followed by constructed, industrial and artificial habitats (J; 14%) and woodland and forest (G; 13%) are most at risk. These results contrast significantly with the ones observed for invasive plants in Spain where most invaders occupy J habitats (61%) followed by agricultural habitats (I; 14%) (Sanz-Elorza et al. 2004; $\chi^2_8$ = 278.37, P<0.0001). Instead, F habitats and woodlands (G) are the least invaded (less than 5%). In fact, Mediterranean woody habitats seem to be resistant to invasion Vilà et al. (2007). In Europe, woody habitats also contain a lower proportion of alien plants than expected from the intensity of propagule pressure (Chytrý et al. 2008).

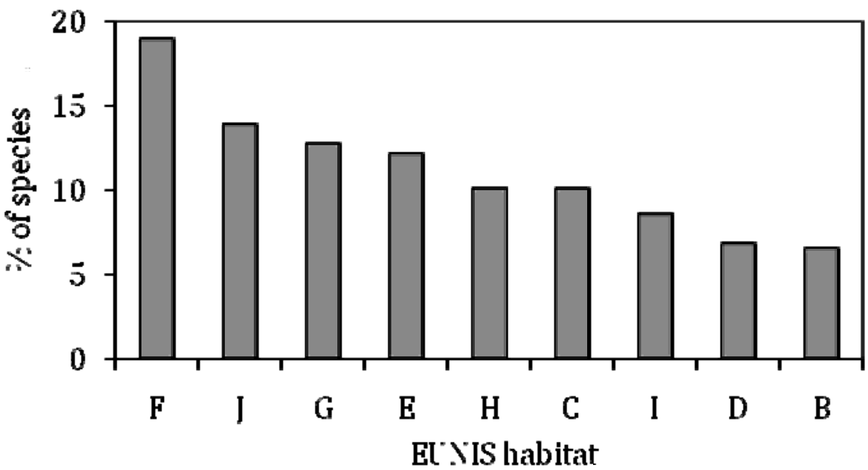



**Figure 3.2** Potential EUNIS habitats (http://eunis.eea.europa.eu/) that could be invaded in Spain by 80 potentially invasive species. **F:** Heathland, scrub and tundra; **J:** Constructed, industrial and other artificial habitats; **G:** Woodland, forest and other wooded land; **E:** Grasslands and lands dominated by forbs, mosses or lichens; **H:** Inland unvegetated or sparsely vegetated habitats; **C:** Inland surface waters; **I:** Regularly or recently cultivated agricultural, horticultural and domestic habitats; **D:** Mires, bogs and fens; **B:** Coastal habitats.

Finally, the hazards posed by these potentially invasive species could be mainly ecological (62%) and socioeconomic (29%). Regarding the economic sectors affected, 50% of the potential invasive plants could impact on the nature conservation sector, 11% on the agricultural sector and 9% on cattle raising (9%).

### *Ranking potential invaders*

All questions of the WG-WRA test could be answer and 82% (40 ± 0.45) of the questions of the WRA, too. Main gaps of knowledge correspond to reproductive characteristics of the screened plants (35%) and to dispersal mechanisms (25%). According to both tests most screened species are potentially invasive in Spain (Appendix III). Nine species, though, showed scores equal or lower than six in the WRA test, which suggest that their invasive potential requires further evaluation.

The species can be divided into three class distributions according to their score (Figure 3.3). Most screened species have intermediate values in both tests (Figure 3.3). In fact, there is a positive relationship between the values from the two risk assessment tests, showing that both tests ranked species in a similar way ($y = 0.78x + 14.16$; $r^2 = 0.97$; Figure 3.4). However, twenty-nine species (37 %) modified their position over the three class distributions; 26 have higher risk class distribution in the WG-WRA than in the WRA and 3 have lower risk in the WG-WRA (Appendix III). Nonetheless, these differences among the positions of the species among the two tests can be explained by the fact that class distributions of the WRA test have been chosen arbitrary and no thresholds have been set up by the authors designing the protocols (Pheloung et al. 1999).

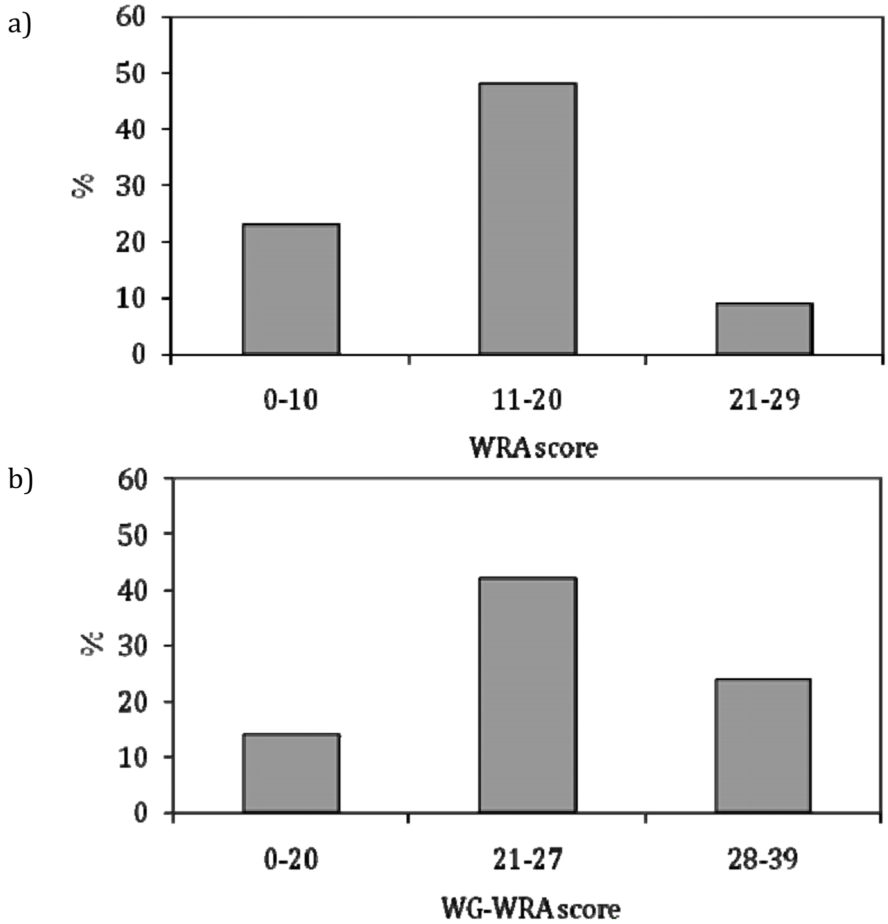



**Figure 3.3** Class distribution of scores of 80 potentially invasive species in Spain according to the WRA (a) and to the WG-WRA system (b).

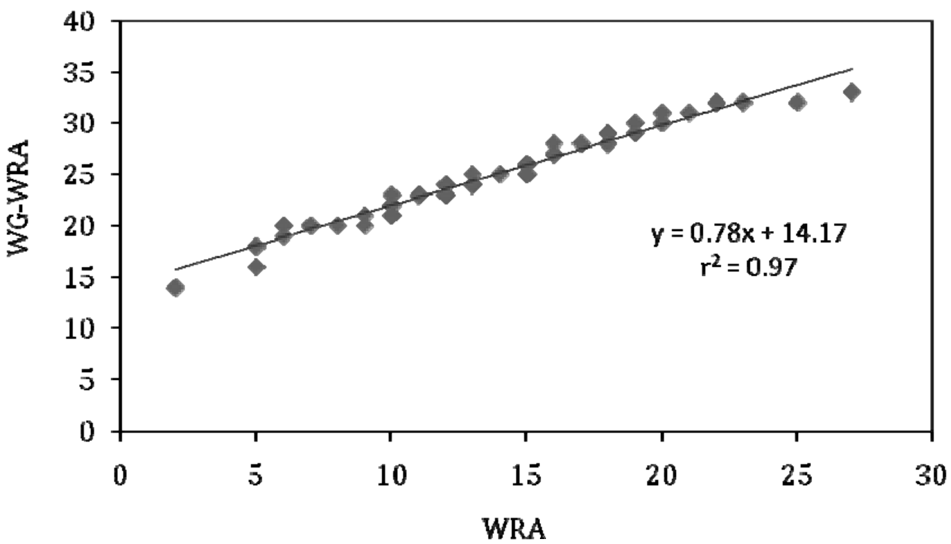



**Figure 3.4** Relationship between the scores of the 80 potentially invasive species for Spain according to their WRA and WG-WRA score.

*Chromolaena odorata* (L.) R.M.King & H.Rob. (Fam. Asteraceae) obtained the highest score in both tests and, therefore, is the species with the highest risk to become invasive in Spain if introduced. *C. odorata* is a fast-growing perennial shrub, native to South America and Central America. It has been introduced mainly as an ornamental plant, but also accidentally, in many tropical regions of Asia, Africa and the Pacific and in the Mediterranean areas of Australia and South Africa, where it is a very invasive weed. Indeed, it is considered one of the 100 worst invasive species of the world by the IUCN (http://www.issg.org/database/welcome/). It forms dense stands that prevent the establishment of other plant species and is an aggressive competitor with allelopathic effects (Onwugbuta – Enyi, 2001). It is also a nuisance weed in agricultural lands and commercial tree plantations, thus, preventive measures should be taken to avoid its naturalization and trade (Roder et al. 1995).

The class distribution with the highest scores include five aquatic species: *Cabomba caroliniana* A. Gray, *Hydrocotyle ranunculoides* L. f. , *Salvinia molesta* D. Mitch, *Ludwigia peploides* (Kunth) P.H. Raven, *Alternanthera philoxeroides* (Mart.) Griseb., confirming their great capacity of invasion, their devastating impacts, not only environmental but also economic, and their complicated and costly control (Pimentel et al. 2005). Aquatic plants are very easy to purchase and there is no control over their use and trade, therefore, early detection efforts should be placed to detect the entrance of these species in the country.

Main ranking differences among species in the two tests can be a consequence of different information required to answer the questionnaires. Both tests give a relevant importance to the distribution of the target species in other regions (e.g., Scott & Panetta 1993) and to climate matching (Bomford 2003, Thuiller et al. 2005). However, only the WRA test incorporates the historical pest status of the target plant in other regions. The invasive-elsewhere criterion is generally considered very important for the assessment of potentially invasive species (Reichard & Hamilton 1997, Rejmanek et al. 2005b).

The potential impacts over agriculture and the existence of weedy congeners are taken into account by both tests but the WRA test also considers other economic impacts such as negative effects on recreation, amenity, tourism, etc. Moreover, the WRA test also allows, by splitting the total score of the WRA, an estimation of whether a plant is more likely to impact on agriculture or on natural ecosystems, which can also assist environmental managers in making recommendations. Almost all potential invasive species screened with the WRA (97.5 %) were regarded as both environmental and agricultural weeds and only 2.5 % were environmental weeds only.

Regarding biological traits, life form, vegetative growth, seed viability, type of reproduction and dispersal mode are included in both tests. While the WRA test give higher risk to aquatic plants and grasses, the WG-WRA test confers also high scores to woody perennial species as woody plants have been reported as successful invaders in central Europe, where the protocol was developed (Krivánek & Pyšek 2006). Undesirable traits like spinescence, toxicity and increment of fire hazard are only incorporated in the WRA test and they fit very well with traits that can have an impact on pasture and fire regime, two of the most relevant features shaping the vegetation of Mediterranean type-ecosystems (Di Castri & Mooney 1973). The WRA also includes evolutionary considerations such as the hybridising potential of the invader, a trait which has been shown to be relevant to reinforce invasion (Vilà et al. 1998). Finally, only the WRA scheme mentions plant persistence attributes such as persistent seed bank or high resprout capacity, which can be an indirect indicator of the management effort required (Hiebert & Stubbendieck 1993, Burch & Zedaker 2003).

In this study we have tested whether invasive species of other Mediterranean regions would become invasive if introduced to Spain. However, another comprehensive list of potential invasive species could be obtained considering all native species to the rest of Mediterranean regions of the world. Given that similarity in climate between native and target regions has long been recognized as a basic requirement for a successful invasion (Scott & Panetta 1993, Bomford et al. 2005, Thuiller et al. 2005), this approach would also be a useful tool to identify potential invasive plants for Spain.

## Conclusions

The rapidly increasing number of alien species and their tremendous costs to the environment and society (Pimentel et al. 2005) urgently require decision-making tools for the management of plant invasions and the regulation of plant trade in order to diminish the risks of new introductions (Sandlund et al. 1999). Risk assessment schemes are valuable tools to diminish the risk of invasion and to concentrate resources on preventing from entering pathways those species with higher risk of invasion. In fact, they are the first step towards developing restrictive policies regarding the introduction of novel species. This does not involve denying the agriculturist and horticulturist the option to provide their customers with new plants. It does mean that introduced plant species would only be allowed to be imported if they have not demonstrated capacity to become invasive. If a species is judged to have high invasive potential, rejection for introduction is the prudent policy. The high cost:benefit ratio associated with invasive species risk assessments (i.e., cost of allowing an invasion vs. benefit of allowing introduction of a presumed non-invasive) strongly recommends to use a very conservative approach. Besides, risk assessment schemes could also be valuable for the development of lists of alien plant species that can be permitted to be imported because it has been demonstrated that they pose a low threat of becoming invasive.

The two risk assessment schemes used in this study identify a wide range of potential invasive species not introduced yet to Spain and that are likely to cause economic and environmental impacts. The fact that two different methods ranked species in a similar way increases the probability to identify invasive plants correctly. The species with higher scores on the risk assessment should be prohibited or kept out of trade in pathways in which species are specifically marketed such as aquarium trade, gardening and seed purchase. However, it is important to take into account that alien plants can be invasive in one habitat of their introduced range but not in another, depending on particular habitat properties such as disturbance regime, microclimate/soil properties or propagule pressure (Thompson et al. 1995, Chytrý et al. 2008). Therefore, besides the listed and ranked species presented in this paper more could be invasive to the country. As a consequence, although policies would need to be implemented at the national level, local analyses within that larger area are also necessary.

## Acknowledgments

We thank N. Gassó, J. Pino and two anonymous reviewers for valuable comments on a previous version of the manuscript. This study has been partially funded by the EC within the FP 6 Integrated Project ALARM (Assessing LArge-scale environmental Risks for biodiversity with tested Methods - GOCE-CT-2003-506675; http://www.alarmproject.net; Settele et al. 2005) and the CONSOLIDER-INGENIO 2010 project: Spanish woodlands and global change: threats and opportunities (MONTES-CSD2008-00040).

# Chapter 4

# Native plant community response to alien plant invasion and removal: a review[4]

[4] Andreu J, Vilà M. Native plant community response to alien plant invasion and removal. *Management of Biological Invasions* (Submitted in October 2011).

## Abstract

Given the potential ecological impacts of invasive species, removal of alien plants has become an important management challenge and a high priority for environmental managers. To consider that a removal effort has been successful requires both, the effective elimination of alien plants and the restoration of the native plant community back to its historical composition and function. We present a conceptual framework based on observational and experimental data that compares invaded, non-invaded and removal sites to quantify invaders' impacts and native plant recover after their removal. We also conduct a meta-analysis to quantitatively evaluate the impacts of plant invaders and the consequences of their removal on the native plant community, across a variety of ecosystems around the world. Our results confirm that invasion by alien plants is responsible for a local decline in native species richness and abundance. Our analysis also provides evidence that after removal, the native vegetation has the potential to recover to a pre-invasion target state. Our review reveals that observational and experimental approaches are rarely used in concert, and that reference sites are scarcely employed to assess native species recovery after removal. However, we believe that comparing invaded, non-invaded and removal sites offer the opportunity to obtain scientific information with relevance for management.

## Introduction

In many parts of the world, some alien plant species are threatening biodiversity by altering native community and ecosystem structure, function and dynamics (Vitousek et al. 1997, Parker et al. 1999, Mack et al. 2000, Ehrenfeld 2010, Powell et al. 2011). In addition, considerable socioeconomic and human welfare impacts have also been reported (Pimentel et al. 2005, Kettunen et al. 2009, Vilà et al. 2010). Given these ecological and socioeconomic impacts, management of alien plants has become an important challenge and a high priority for the conservation of native species and natural areas (Zavaleta et al. 2001, Smith et al. 2006, Swab et al. 2008).

In many natural areas invaded by alien plants, management practices include the removal of alien plants to reduce their population size or, if possible, to eradicate them (Manchester & Bullock 2000, D'Antonio & Meyerson 2002, Price & Weltzin 2003, Holmes et al. 2005). However, to consider that a removal effort has been successful requires both, the effective elimination of alien plants and the restoration of the native plant species community back to its historical composition and function (Zavaleta et al. 2001, SER 2002, Hulme 2006). In order to effectively control invasions, an assessment of the magnitude of their impacts is also required (Stinson et al. 2007). Thus, a comprehensive understanding of the success of alien plant management, would ideally involve observational and experimental comparative studies between invaded, non-invaded and removal sites, that asses the impacts of the alien plant and the resulting native species assemblage, after its removal.

Basically, three different types of comparisons can inform us on the magnitude and direction of changes of the native plant community with invasion or the removal of a particular invasive species (Figure 4.1). Each comparative approach will provide information about the invasion and the removal success and will give answer to different research questions or hypothesis. So far, there are many observational studies comparing invaded communities to their non-invaded reference counterparts (Levine et al. 2003; Comparison A in Figure 4.1). This approach has been used to infer alien species impact on the native community such as reduced plant richness or diversity, or reduced seedling recruitment (Gaertner et al. 2009). However, it does not demonstrate causality as the observed outcome can be confounded with site differences between invaded and non-invaded plots.

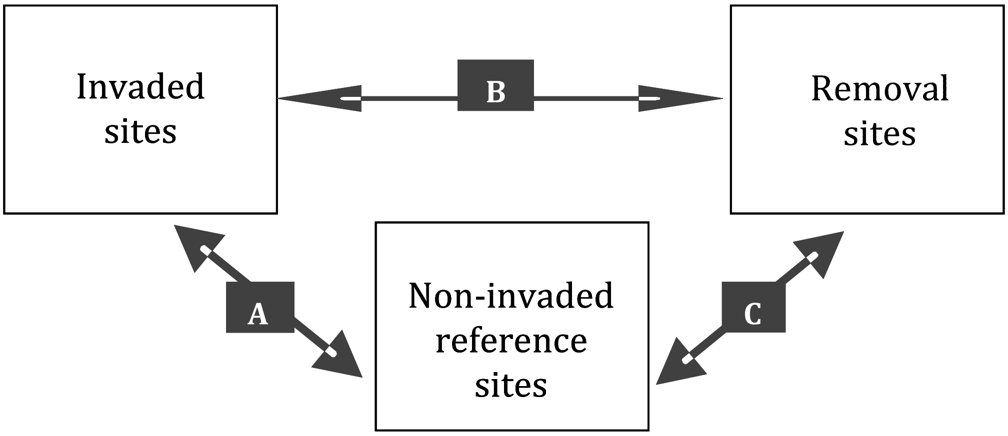



**Figure 4.1** Schematic diagram that summarize the different types of comparison that can be used to assess the impact of an invader and the recovery of the native community after alien plant removal. (A) Observational approach, can provide a potential assessment of impacts (B) Experimental approach, can provide a causal assessment of impact, and (C) Experimental approach, provides an assessment of the native community recovery after removal of the invader.

Other studies have conducted field removal experiments to eliminate the invader and compare the sites where the invasive species have been removed with invaded (unmanaged) sites (e.g., Ogle et al. 2000, Morrison 2002, Hejda & Pyšek 2006, Hulme & Bremner 2006; Comparison B in Figure 4.1). These comparisons present substantial opportunities for demonstrating the impact of plant invasions on the native community (Díaz et al. 2003). If the invasive species is competing with native species, we expect higher richness in removal sites than in invaded ones, given that the native vegetation would be released from the use of resources by the invader. However, the outcomes of these comparisons can be confounded with short-term, local scale disturbance effects after removal. In general, these studies are purely experimental to test impact and do not have a management perspective because they do not measure the success of the removal effort.

Many times removal of alien species alone does not always lead to the reestablishment of the desirable native communities because land-use history, seed bank availability and disturbance regime can all strongly influence the outcome of the removal effort, which might not accomplish the preferred level of recovery (Partel et al. 1998, Zavaleta et al. 2001). Thus, an increasingly common goal of ecosystem restoration is to assess whether removal sites may achieve the high levels of plant species, trait and functional group diversity found in remnant intact sites (Mason & French 2007, Stinson et al. 2007, Truscott et al. 2008). In this sense, the identification of non-invaded and non-managed reference sites and their comparison with removal sites may be crucial to assess the success of the removal effort (Chapman & Underwood 2000; Comparison C in Figure 4.1). A successful removal strategy would be the one in which vegetation recovers similarly to a reference community in species composition and richness (McCoy & Mushinsky 2002). This type of comparisons can be also valuable to evaluate eventual negative side-effects of removal techniques such as the proliferation of other alien species or soil erosion disturbances (Álvarez & Cushman 2002, Truscott et al. 2008).

In this study, we conducted a global literature review to quantitatively evaluate the impacts of invaders and the consequences of their removal on the native plant community, across a variety of ecosystems around the world. The response of the native plant community to removal may be an indicator of the success of the management efforts. We evaluate the magnitude and direction of change of two response variables, native plant species richness and abundance, to alien plant invasion and removal, using a meta-analytical approach (Rosenberg et al. 2000). We focused on native species richness and abundance as indicators of community level response because these are the most commonly used response variables across all reviewed papers. Meta-analysis is a technique of quantitative research synthesis that can provide a more robust method than traditional story-telling or vote-counting literature review. It also provides the opportunity to explore heterogeneity among studies and identify large-scale patterns across species and geographic regions (Steward 2010, Harrison 2011).

The framework described in Figure 4.1 is used as a guide to answer the following questions: (1) Does alien plant invasion decrease native species richness and abundance (Comparison A)? (2) Do alien plant removal increase native species richness and abundance (Comparison B)? and finally, (3) Does removed sites resemble to reference non-invaded sites (Comparison C)?

## Methods

### *Literature search*

To compile quantitative evidence of the response of the recipient plant community to alien plant invasion and removal, we searched for papers on the ISI Web of Knowledge (www.isiwebofknowledge.com) database on July 2009, with no restriction on publication year, using the following search term combinations: (Invasive plant OR non-native plant OR exotic plant OR alien plant) AND (restor* OR removal OR clear* OR success OR response OR rehabilitat*) AND (uninvaded OR non-invaded) OR (impact* OR invasion OR effect). Additional literature was obtained screening the reference lists from all retrieved papers. An initial set of 120 studies was evaluated in order to assess their potential for meeting the selection criteria for inclusion in the review. Studies that did not meet these criteria were omitted. The first criterion was that the focus of the study should be a particular terrestrial or riparian alien plant species, excluding all papers involving strict aquatic species. Second, we decided to include only studies in which native richness or native abundance (i.e., native cover, native density and native biomass) after invasion or removal was assessed. Studies reporting on other community indicators such as diversity, evenness, alien richness and abundance, were too scarce for some comparisons. We also rejected articles dealing with total species richness, total diversity or total abundance, because our main goal was to focus on the response of the native community. Finally, for those articles involving alien plant removal, we discarded chemical and biological control managements; therefore, we confined our selection of studies to mechanical or manual removal techniques.

### *Data extraction*

A total of 39 published papers involving 34 taxa met our criteria (Appendix IV). Some studies investigated several species, at different sites or at different habitats, what left us with 132 cases for the meta-analysis. Eighty-seven of these cases compared native richness or abundance between non-invaded and invaded sites, 33 between removal and invaded sites and 12 between removal and non-invaded sites.

Among the alien plant species evaluated, perennial herbs (45 cases), shrubs (33 cases) and annual grasses (24 cases) were more often represented than the other life-forms. Most of the studies (36%) have been conducted in North America (36%), Europe (33%) and

Australia (24%), being Mediterranean (34%) and temperate regions (30%) the most investigated (Appendix IV).

For each response variable (i.e., native plant species richness and abundance) we recorded sample size (N), mean ($\overline{X}$) and statistical variation (usually SE or SD) in invaded, non-invaded and removal plots for each study. These data were extracted directly from tables or from graphs using the DATATHIEF II software (B. Thumers; http://www.datathief.org) or, in some cases, also by measuring mean and statistical variation 'manually' using a ruler. For other studies, we obtained data directly from the corresponding authors.

## *Meta-analysis*

As a unit of analysis (i.e., the unit for calculation of effect sizes and their variances), we used pairs of plots of the following comparisons: non-invaded versus invaded (Comparison A; Figure 4.1), invaded versus removal (Comaprison B; Figure 4.1), removal versus non-invaded (Comparison C; Figure 4.1). For each entry of the dataset we calculated the natural log of the response ratio (Ln R) as a measure of the magnitude and direction of the effect size. Ln R is the ratio of a variable in the experimental (e) group to that of the control (c) group (Rosenberg et al. 2000). From each pair of mean values ($\overline{X}$) the effect size was calculated as:

$$\ln R = \ln\left(\frac{\overline{X}^e}{\overline{X}^c}\right)$$

where e is the experimental group and c is the control group (see below for selection of experimental and control groups). The variance of Ln R, $V_{LnR}$ was calculated as:

$$v_{\ln R} = \frac{(s^e)^2}{N^e\left(\overline{X}^e\right)^2} + \frac{(s^c)^2}{N^c\left(\overline{X}^c\right)^2}$$

where *S* is the pooled standard deviation and N is the number of replicates per treatment.

Ln R is a unit-free index which ranges from - ∞ to + ∞ and estimates the effect size as a proportional change. As in classical statistical analysis, the highest effect sizes are from those studies showing large differences between control (i.e non-invaded) and experimental (i.e., invaded or removal) plots and when there is low variability within plots. Zero Ln R values imply no difference in the variable measured between control and experimental plots; positive and negative Ln R values imply a general trend following treatment (invasion or removal) for an increase and decrease, respectively. Ln R calculations and statistical analysis were conducted with the MetaWin v2.1 Software (Rosenberg et al. 2000).

The following effect sizes for native species richness and abundance were calculated:

- Comparison A: non-invaded (control) vs. invaded (experimental) plots
- Comparison B: removal (control) vs. invaded (experimental) plots
- Comparison C: non-invaded (control) vs. removal (experimental) plots.

We tested whether effects sizes across studies were homogeneous, using the $Q_{total}$ statistic based on a chi-squared test. A significant $Q_{total}$ indicates that the variance among effect sizes is greater than that expected by sampling error alone (i.e., effect sizes are not equal across studies).

For each grouping category, we calculated the cumulative effect size (R+) and the 95% CI across the sample of studies, with information on the response variable. Data were analysed using random-effects models ($P_{random}$) which are preferable in ecological data synthesis because their assumptions are more likely to be satisfied (Rosenberg et al. 2000). A cumulative effect size (R+) is considered significant (i.e., no change with invasion or removal) when its 95% CI do not overlap zero. Confidence intervals were calculated using bias-corrected bootstrap resampling procedures (Adams et al. 1997, Rosenberg et al. 2000). The mean percentage of change in a response variable was estimated as ($e^{R+}$ -1) x 100 (Table 4.1).

## Results and discussion

In general, across all studies, results were quite homogeneous as indicated by p-values associated to $Q_{total}$ higher than 0.05 in most cases (Table 4.1). Therefore, the magnitude and direction of the effect sizes did not vary significantly across studies, making the generality of the results highly consistent.

The meta-analysis revealed an overall significant decline of native species richness and abundance after invasion. Invaded plots had lower native species richness and abundance than adjacent reference non-invaded plots (Figure 4.2, Table 4.1). The same trend was found when comparing invaded vs. removal plots (Figure 4.2, Table 4.1). On average, invaded plots contained 23% fewer native species than non-invaded plots and 30% less species than removal plots (Table 4.1). Invaded plots were 41% less abundant in native species than non-invaded plots and 30% less abundant than in removal plots (Table 4.1). For none of the response variables evaluated and none of these comparisons, the CI of the mean effect size overlapped zero (Figure 4.2). Therefore, our review supports that alien plant species has a negative impact on species richness and abundance by replacing native species in the communities they invade (Vilà et al. 2011). Nonetheless, further research should concentrate on the mechanisms underlying alien plant invasions in order to investigate which are the factors that are ultimately responsible for this decline of native species richness and abundance (Levine et al. 2003).

**Table 4.1** Total heterogeneity ($Q_t$) with indication of *P*-values, mean effect sizes (R+), degrees of freedom (df) and 95%-CI for each response variable and type of comparison. Significant results of $Q_t$ are in **bold.** The percentage of change of each response variable with invasion or removal is also indicated. See text for a detailed description of statistical analysis.

| Type of comparison | Response variable | $Q_t$ | *P*-value | R+ | df | 95%-CI | % of change |
|---|---|---|---|---|---|---|---|
| Invaded vs. non-invaded | Native richness | 73.40 | **0.008** | -0.267 | 47 | -0.363 to -0.171 | -23.45 |
| | Native abundance | 46.93 | 0.152 | -0.524 | 38 | -0.756 to -0.292 | -40.77 |
| Invaded vs. removal | Native richness | 9.60 | 0.651 | -0.354 | 12 | -0.557 to -0.152 | -29.83 |
| | Native abundance | 89.48 | **0.000** | -0.361 | 19 | -0.515 to -0.207 | -30.32 |
| Removal vs. non-invaded | Native richness | 9.43 | 0.223 | 0.005 | 7 | -0.189 to 0.198 | 0.47 |
| | Native abundance | 9.03 | 0.108 | -0.007 | 5 | -0.563 to 0.549 | -0.68 |

Moreover, the CI for each variable overlapped between Comparison A and B suggesting that the magnitude of the impact is not significantly different between observational and removal studies. That is, whether the control plot is a non-invaded community or a community where the invader has been removed does not have an influence on the magnitude of the impact. This result indicates that although observational studies do not demonstrate causality they can be used as surrogates to test for impact. The extended time frame associated with observational studies counterbalances the logistic and man-power requirements for removal experiments, and the usually short-term monitoring afterwards (Sol et al. 2008).

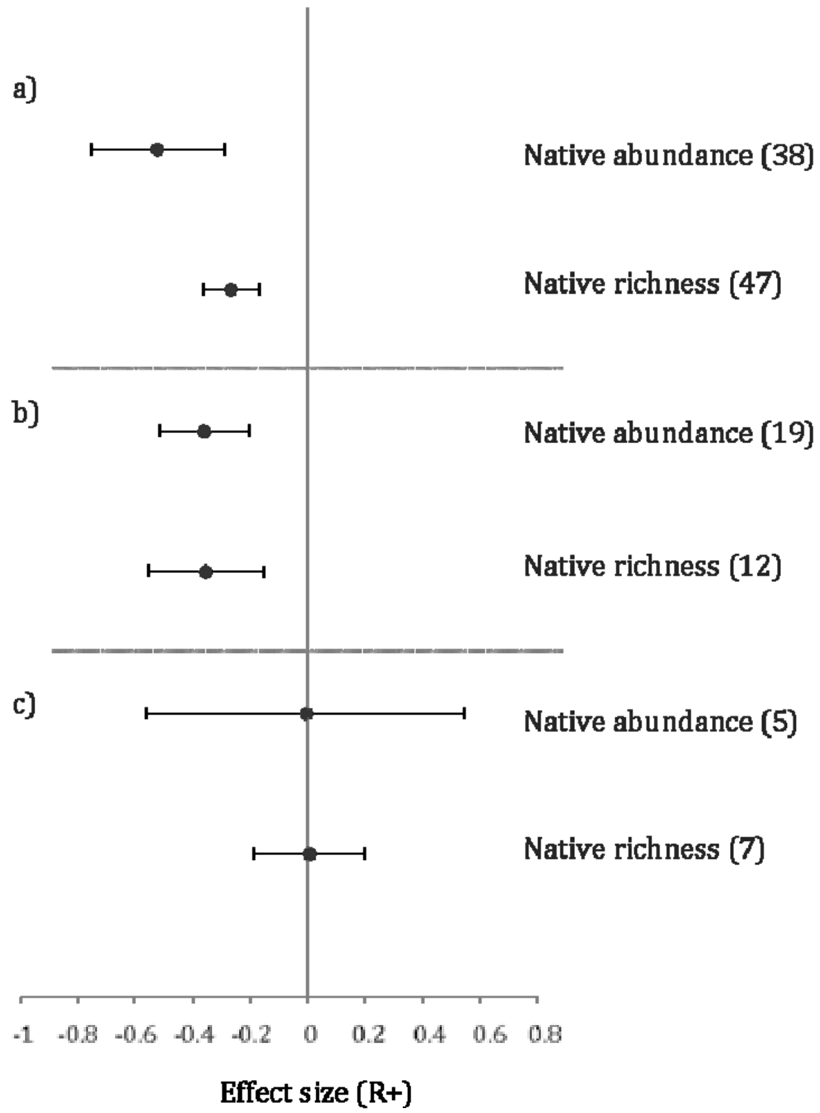



**Figure 4.2** Mean effect size (R+) of differences in native species richness and abundance between a) invaded versus non-invaded, b) invaded versus removal and c) removal versus non-invaded plots. These comparisons are described in Figure 4.1. The bars around the means denote bias-corrected 95%-bootstrap confidence intervals. A mean effect size is significantly different from zero when its 95% confidence interval do not overlap zero. Positive and negative mean effect sizes indicate, respectively, an increase or decrease of the response variable following treatment (invasion in comparisons A and B, and removal in comparisons C). The sample sizes are given next to each variable.

When comparing removal and non-invaded sites we found that after removal, native species richness and abundance reached similar values than in reference non-invaded plots as indicated by R+ values close to zero and the CI of the mean effect size overlapping zero (Figure 4.2, Table 4.1). The magnitude and direction of the native plant community response following removal indicates that restoration of the vegetation to a pre-invasion state is feasible (Truscott et al. 2008). However, one of the consequences of invasive species removal may be to facilitate the proliferation of other invasive species (Álvarez & Cushman 2002, Ogden & Rejmánek 2005, Hulme & Bremner 2006, Truscott et al. 2008), and to cause soil and vegetation disturbance (D'Antonio et al. 1998, Zavaleta et al. 2001). Thus, although removal increases species richness bringing the native plant community closer to non-invaded sites, an increase in the occurrence and abundance of another alien undesired species may also occur and should be assessed. None of these side-effects could have been evaluated in this review because in the vast majority of reviewed papers information about soil and vegetation disturbance was not available and only 3 cases (Holmes et al. 2000, Mason & French 2007, Blanchard & Holmes 2008) specified non-native richness and abundance in removal and non-invaded reference sites, which resulted in the exclusion of these variables from the analysis. Nonetheless, the effective re-establishment of native species suggested in this review, may increase the resistance of the site to re-invasion or to colonization by other alien species.

Despite the complementarily between the three comparisons, they have been rarely used in concert (only 6 papers). In fact, we found only 12 cases in the literature meeting our inclusion criteria, which compared removal sites with non-invaded reference sites. This can be explained by the fact that, unfortunately, alien species are often so widespread by the time they are noticed that a practical limitation of these comparisons is that such comparable non-invaded sites may be extremely rare or in some cases simply impossible to find.

Since removing invasive species requires tremendous time and effort, the potential costs and benefits of managing invaders should be assessed to better inform management and restoration decisions. Thus, it is imperative to determine which invaders have larger effects and to understand the conditions under which restoration projects are likely to succeed. Post-removal monitoring, on both the target invasive species and the invaded community, is extremely helpful because it allows documenting the success of control and provides the opportunity to restrain negative effects before they become severe (Zavaleta et al. 2001, Andreu et al. 2010). Using removal methods without thoroughly testing their effectiveness and non-target effects can lead to routine implementation of inappropriate techniques. Moreover, it is crucial to evaluate if the native vegetation is recovering in order to be able to determine whether further intervention is necessary such as active post-removal revegetation plans. The extent to which post-removal revegetation is required depends on the biology of the invasive species, the magnitude of the invasion and the length of time that the invasive species have been present (D'Antonio & Meyerson 2002).

We think that the conceptual framework presented here (Figure 4.1) may provide guidance for managers and can optimize restoration success. A holistic approach, based on observational and experimental studies, that evaluates both, the influence of invaders and the consequences of their removal, should be more frequently used in concert. This approach may allow managers to gain a more complete picture of the magnitude and direction of changes after its control measures, giving them an idea of the impact caused by an invasive species as well as the capacity of the native community to achieve the desired target state. The use of this comparative monitoring poses challenges to ecologists in deciding how to choose reference sites, how to select response variables, and how to sample in a way that minimizes the influence of confounding factors (Wilkins et al. 2003).

## Conclusions

We have presented a conceptual framework based on observational and experimental data that compare invaded, non-invaded and removal sites to quantify invaders' impacts and native plant recover after their removal. We have quantitatively reviewed the impacts of alien plant invasions on native plant community and its response to their removal. Our results have confirmed that the invasion by alien plants is responsible for a marked local decline in native species richness and abundance, posing significant impacts to the native community. On the other hand, our quantitative approach also provides evidence that after removal, native vegetation has the potential to recover to a pre-invasion target state. Moreover, our review revealed that observational and experimental approaches are rarely used in concert, and that reference sites are scarcely employed to assess native species recovery after removal. Since the success of any alien removal operation depends, ultimately, on this recovery we believe that the comparative approach presented here is crucial because it can empirically answer fundamental ecological questions about how invasions affect native communities. Moreover, it might simultaneously demonstrate how well a given removal strategy can achieve specific management goals. Therefore, we suggest that this comparative approach can make restoration efforts involving alien plant species more practical and successful, supporting science-based management decisions for the protection and restoration of biological diversity.

## Acknowledgments

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# Chapter 5

# Vegetation response after removal of the invasive *Carpobrotus* hybrid complex in Andalucía, Spain[5]

[5] Andreu J, Manzano-Piedras E, Bartomeus I, Dana ED, Vilà M (2010) Vegetation response after removal of the invasive *Carpobrotus* hybrid complex in Andalucía, Spain. *Ecological Restoration* 28: 440-448.

## Abstract

We evaluated the ecological success of the manual removal of *Carpobrotus* species, a putative hybrid complex of a South African perennial mat-forming plant, by comparing treated, non-invaded, and invaded plots across coastal Andalucía in southern Spain. As a measure of the management effectiveness, we quantified the density of *Carpobrotus* seedlings and resprouts in treated plots one year after treatment. Response of the plant community to removal was assessed by comparing native species richness, cover, diversity, and species composition among treatments. Removal greatly reduced to a great extent *Carpobrotus* density. However, successful control will require repeated hand-pulling treatments. Treated plots had a significant increase in species richness, especially annual plants, compared to invaded plots, but both had the same native plant cover and diversity. We found similar species composition between removal and non-invaded plots, indicating that revegetation is not necessary. Long-term monitoring is necessary to determine whether these observed patterns of community response are transient or stable through succession.

## Introduction

Given the potential negative impact of alien plants on native species, ecosystems, and human health and economies (Mack et al. 2000, McKinney & Lockwood 1999, McNeely 2001, Pimentel et al. 2005), control of invasive species has become an important challenge for land managers, as well as a common component of restoration efforts (Zavaleta et al. 2001, Smith et al. 2006). Since removing invasive species requires tremendous time and effort, the potential costs and benefits of managing invaders should be assessed. Such evaluations need to include measures of plant community response, not just such factors as funding and degree of community commitment.

The invasive South African succulent genus *Carpobrotus* (Aizoaceae) is an example of an alien plant that often dominates plant communities in many Mediterranean regions of the world (D'Antonio & Mahall 1991, D'Antonio 1993, D'Antonio et al. 1993, Vilà et al. 2006). *Carpobrotus* species are crawling, mat-forming, succulent chamephytes (plants with buds near ground level, Raunkier 1977) easily recognized by their succulent, finger-shaped, and triangular-section leaves (D'Antonio 1990, D'Antonio 1993). Introgressive hybridization is very common (Vilà et al. 1998), occurring throughout coastal California between the non-native hottentot fig (*Carpobrotus edulis*) and the putative native sea fig (*C. chilensis*), leading to a high abundance of invasive hybrid morphotypes that compete aggressively with native plant coastal species (D'Antonio 1990, Albert et al. 1997, Vilà & D'Antonio 1998). In Spain, *Carpobrotus* species, known locally as *uña de gato*, may be hybrids between hottentot fig and Sally-my-handsome (*C. acinaciformis*).

*Carpobrotus* forms large mats on coastal rocks, cliffs, and sand dunes owing to its profuse clonal growth and long-distance dispersal by vertebrates (D'Antonio 1990, D'Antonio 1993, Traveset et al. 2008). The fleshy fruits bear a large number, often over a thousand, of small seeds (Bartomeus & Vilà 2009) that are eaten and widely dispersed by several mammals such as rabbits (D'Antonio 1990) and rats (Bourgeois et al. 2005). *Carpobrotus* has a long-lived seed bank that can remain viable in the soil for at least two years (D'Antonio 1990). *Carpobrotus* is able to grow from multiple axes, rooting where nodes contact the soil, and spreads radially at rates as high as one meter per year (Wisura & Glen 1993). It competes aggressively with native plant species, achieving high rates of space colonization, which suppresses the growth and establishment of other plants (D'Antonio & Mahall 1991, Albert 1995, Suehs et al. 2004, Vilà et al. 2006). Furthermore, it also interacts indirectly with native vegetation by altering soil chemistry (Conser & Connor 2009).

It was first introduced as an ornamental plant into Europe around the 17th century at the Leyden Botanical Garden, the Netherlands, and since then it has been cultivated in other European botanical gardens (Fournier 1952). However, the progressive expansion and naturalization in the Mediterranean Basin started in the beginning of the 20th century (Sanz-Elorza et al. 2004). Nowadays, it is considered invasive in Europe (Albania, France, southern UK, Portugal, Italy, Greece, Montenegro, and the Canaries and other Mediterranean islands), northern Africa, southern Australia, New Zealand, and USA (California and Florida) (Sanz-Elorza et al. 2004).

In Spain, it was introduced intentionally for gardening, landscaping, and dune stabilization in the beginning of the 20th century, owing to its fast clonal growth, low water requirements, and showy, large flowers (Sanz-Elorza et al. 2004). Due to the large extent of invaded areas in coastal communities, it is one of the most costly invasive species in Spain (Andreu et al. 2009). We conducted a regional survey to test the short-term effectiveness of *Carpobrotus* removal and native vegetation response in coastal sand dunes across Andalucía (southern Spain). In particular, we addressed the following two questions: 1) Has *Carpobrotus* been successfully controlled in our study sites one year after treatment? 2) Are native species richness, cover, diversity, and composition after *Carpobrotus* removal similar to reference communities?

## Study sites and experimental design

The study was conducted in six coastal localities in four provinces of Andalucía where *Carpobrotus* had been removed the year before as part of the Andalusian Plan for Control of Invasive Species (Ortega Alegre & Ceballos 2006) (Figure 5.1). Overall, 300 ha had been treated by hand-pulling *Carpobrotus*. A total of 400 T of plant material was transferred to compost areas. This plant is readily cloned by rooting stem fragments that contain just one node; thus it was crucial to fully remove all individuals and also any buried stems (D'Antonio 1990).


**Figure 5.1** Six experimental sites where *Carpobrotus* was removed and monitored in Andalucía (southern Spain): a) Parador Mazagón (Huelva); b) Punta Camarinal (Cádiz); c) Torrelapeña (Cádiz); d) Artola 1 and 2 (Málaga); and e) Punta Entinas (Almería).

These localities provided a representative sample of the entire Andalusian coast and the typical habitat types where *Carpobrotus* invades worldwide. The vegetation in the study sites is typical Mediterranean coastal shrublands dominated by chamephytes. The main species are European beachgrass (*Ammophila arenaria*, Poaceae), mastic tree (*Pistacia lentiscus*, Anacardiaceae), salvia cistus (*Cistus salviifolius*, Cistaceae), *Silene nicaeensis* (Caryophyllaceae), sweet alyssum (*Lobularia maritima*, Brassicaceae), silver sea stock (*Malcolmia littorea*, Brassicaceae), *Echium gaditanum* (Boraginaceae), and *Anacyclus radiatus* (Asteraceae). The climate is Mediterranean, with warm dry summers and mild, wet winters.

In order to determine effectiveness and vegetation responses to removal treatments, we established the following three treatments: 1) control plots ($n$ = 18) where *Carpobrotus* was not present and there was no history of invasion and vegetation removal management; 2) invaded plots ($n$ = 13) with high *Carpobrotus* cover (circa 70%) and no history of removal; and 3) treated plots ($n$ = 46) where *Carpobrotus* was removed. The distance between plots within a locality ranged from 10 m to 50 m. This approach allowed us to distinguish changes caused by removal, since observing treatment plots over time would not allow differentiation of treatment effects from changes due to natural fluxes

(Swab et al. 2008). Table 5.1 provides details for all six localities about location, habitat, most common plant species, and number and size of plots for each treatment.

The location, number, and size of experimental plots were decided with the aid of land managers one year after *Carpobrotus* removal. Owing to the idiosyncrasies of the removal treatments, caused mainly by differences in the size and spatial position of *Carpobrotus* patches and vegetation structure, plot sizes were not identical between sites. Plot sizes ranged from 2.5 × 2.5 m (6.25 $m^2$) to 10 × 10 m (100 $m^2$).

As a measure of the management effectiveness, density of *Carpobrotus* seedlings or resprouts (hereafter recruits) was determined within each plot as the number of recruits per square meter. *Carpobrotus* recruits were classified depending on their number of leaves in 4 different categories based on increasing branching patterns (<10 leaves, 10–24 leaves, 25-49 leaves, and >50 leaves). Recruits with fewer than 10 leaves were probably seedlings, and those with more than 50 leaves were probably remnants left in place unintentionally when removing the species.

Vegetation response to *Carpobrotus* removal was measured by the point-intercept method. Plant surveys were carried out at 20 cm intervals along the perimeter of the experimental plot by recording all plant species contacting an imaginary vertical line at each interval point. In each plot, we identified all species at 300-1,000 points, depending on the size of the plot. This method provided a record of species composition and abundance in each plot, a measure of plant cover, and, indirectly, an estimation of species richness and diversity.

Most taxa were identified to the species level, with grasses (Poaceae) being pooled, and then labelled as native or non-native and assigned to one of the five Raunkier (1977) plant life-forms. This plant classification system is based on the position of perennating buds in relation to the soil surface: chamephytes, geophytes (survival via underground food-storage organs such as rhizomes, tubers, or bulbs), hemicryptophytes (perennating buds at ground level and aerial shoots dieback), phanerophytes (perennating buds or shoot apices on aerial shoots) and therophytes (survival as seeds—annual or ephemeral plants).

We calculated the relative cover of individual plant species as the proportion of contacts of each species relative to the total number of plant contacts per transect, followed by the relative cover of alien plants and each Raunkier life-form. Total native vegetation cover was calculated as the total number of contacts of all native species in relation to total number of contacts including bare ground, *Carpobrotus* and other alien species, if present. We measured the species richness of natives and non-natives. Native diversity was calculated using the Shannon–Wiener diversity index (*H*), which is sensitive to rare species. All of these response variables were compared between control, invaded, and treated plots.

In order to determine whether, within a site, treated plots exhibited the same species composition as control plots, we calculated the Sorensen Similarity Index (*S*). This index ranges between 1 (same species composition) and 0 (most varied species composition).

**Table 5.1** Site information for all six localities in an experiment to control *Carpobrotus* in coastal vegetation of Andalucía, Spain. Raunkiaer life-form of each species is indicated in parenthesis: P = Phanerophyte, T = Therophyte, C = Chamephyte, G = Geophyte, H = Hemicryptophyte.

| Locality (Province) | Habitat | Native Species | Number of Plots (size, in m) Invaded | Non-invaded (Control) | Treated |
|---|---|---|---|---|---|
| Punta Camarinal (Cádiz) | Rock shores, sea-cliffs, and stable dunes | *Anacyclus clavatus* (T)<br>*Armeria pungens* (H)<br>*Cyperus capitatus* (G)<br>*Euphorbia terracina* (C) | 5 (10 × 10) | 0 | 5 (10 × 10) |
| Torrelapeña (Cádiz) | Shifting and stable dunes | *Ammophila arenaria* (H)<br>*Lotus creticus* (C)<br>*Malcolmia littorea* (C)<br>*Medicago littoralis* (T) | 5 (5 × 5) | 0 | 5 (5 × 5) |
| Artola 1 (Málaga) | Shifting and stable dunes | *Helichrysum stoechas* (C)<br>*Lotus creticus* (C)<br>*Ononis natrix* (C)<br>*Pistacia lentiscus* (P) | 0 | 5 (5 × 5) | 5 (5 × 5) |
| Artola 2 (Málaga) | Shifting dunes | *Cynodon dactylon* (H)<br>*Lotus creticus* (C)<br>*Medicago littoralis* (T)<br>*Pancratium maritimum* (G) | 3 (5 × 5) | 3 (5 × 5) | 3 (5 × 5) |
| Mazagón (Huelva) | Stable dunes | *Cistus salviifolius* (P)<br>*Medicago littoralis* (T)<br>*Paronychia argentea* (C)<br>*Rumex tingitanus* (C) | 0 | 5 (5 × 5) | 10 (2.5 × 2.5) |
| Punta Entinas (Almería) | Stable dunes | *Cyperus capitatus* (G)<br>*Helichrysum stoechas* (C)<br>*Phagnalon saxatile* (C)<br>*Reichardia tingitana* (T) | 0 | 5 (2 × 10) | 5 (2 × 10) |

## Data analysis

As data did not fit parametric test assumptions, differences in the density of the leaf categories of *Carpobrotus* recruits were analyzed with Kruskal-Wallis tests. We also performed a multiple comparison test after Kruskal-Wallis to assess differences in leaf categories using the pgirmess package and the Kruskalmc procedure of R (vers. 2.6.2, R Foundation for Statistical Computing, Vienna, Austria).

Plot sizes differed between sites. Since species richness is dependent on the number of specimens counted and, therefore, on sample size, we transformed observed species numbers to expected values for a given sampling size by rarefaction curves (Sanders 1968, Hurlbert 1971, Gotelli & Colwell 2001) using the rarefy function in the vegan package of R. Rarefaction curves standardized the sampling in each of the plots to the minimum sample size ($n$ = 300), which permits us to use rarefied data to make species richness and diversity comparisons (Gotelli & Colwell 2001). All analyses were done with rarefied data.

Some response variables could not be normalized with data transformation. Therefore, differences in native species cover, richness, diversity, and functional group cover between treatments (invaded, control, and treated) were tested with a generalized linear mixed model (GLMM) with a Poisson distribution of errors and a logit link function (Crawley 2002). The logit link function ensures that all the fitted values are positive, while Poisson errors are recommended to deal with integer (count) variables, which are often right-skewed and have variances that are equal to their means (Crawley 2002). Treatment was included as the fixed factor, and site was included as a random factor to account for spatial autocorrelation. Models were run using the glmmPQL procedure of the MASS package in R. We also calculated the power of our analysis (ß) to assess the probabilities of Type II error, given our small sample size. In order to test if *Carpobrotus* removal increased colonization by other alien species, we compared alien species cover among treated, control, and invaded plots using the Kruskal-Wallis test. Mean values ± standard errors are given throughout the text.

# Results and discussion

## *Effectiveness of Carpobrotus removal*

Recruit density in treated plots across sites averaged 0.13 ± 0.09 recruits per square meter. No reestablishment of *Carpobrotus* occurred in 52% of the treated plots. One of the sampled sites (Punta Camarinal) accounted for most of the observed *Carpobrotus* recruits (63%). Recruits with fewer than ten leaves were significantly more abundant than recruits with at least 10 leaves (Figure 5.2).

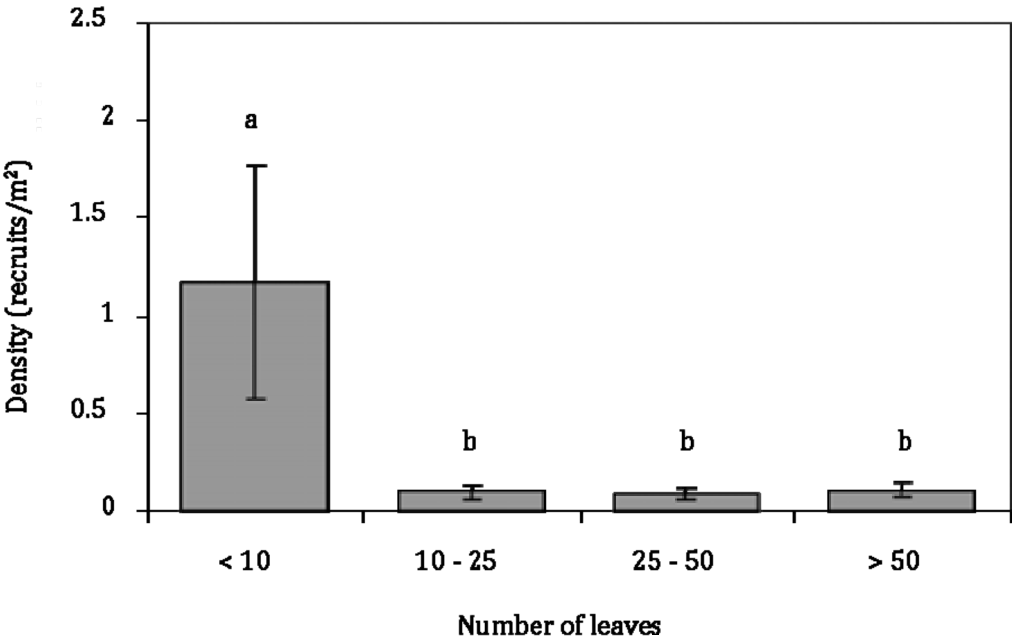


**Figure 5.2** Mean (± SE) *Carpobrotus* recruit density one year after removal from six coastal sites in Andalucía, Spain, grouped by the number of leaves per recruit. Different letters indicate significantly different values ($H$ = 8.74, df = 3, $P$ = 0.032).

Low densities of *Carpobrotus* recruits one year after treatment indicated that short-term management had considerably reduced *Carpobrotus* presence, although it had not eradicated the species. Recruits with fewer than 10 leaves, probably seedlings, were the most abundant, which suggest the importance of the seed bank in the reestablishment capacity of *Carpobrotus*.

### *Native plant species cover, richness and diversity*

The GLMM model revealed significantly higher values of rarefied species richness in treated plots (7.20 ± 0.40) than in invaded plots (6.64 ± 0.29; Figure 5.3a), indicating that *Carpobrotus* may have an impact on species richness by replacing native species in the communities it invades (Brandon et al. 2004, Hejda & Pyšek 2006, Hulme & Bremner 2006). However, we found no significant differences in total native species cover (Figure 5.3b) and diversity (Figure 5.3c). These results are consistent with other case studies (Ogden & Rejmánek 2005, Vidrà et al. 2007, Swab et al. 2008, Pavlovic at al. 2009) and probably are due to a low abundance of new recruits and the short-term scale of our study. Neither did we find significant differences between treated and control plots with respect to plant cover, richness, and diversity, indicating that regeneration after *Carpobrotus* removal results in coastal dune communities similar to reference native communities. Nonetheless, we have to be cautious when interpreting these results, mostly in the case of total native species cover between treated and invaded plots ($p$ = 0.07); the limited power of our statistical analysis (ß = 0.67) could prevent us from detecting possible significant differences among treatments.

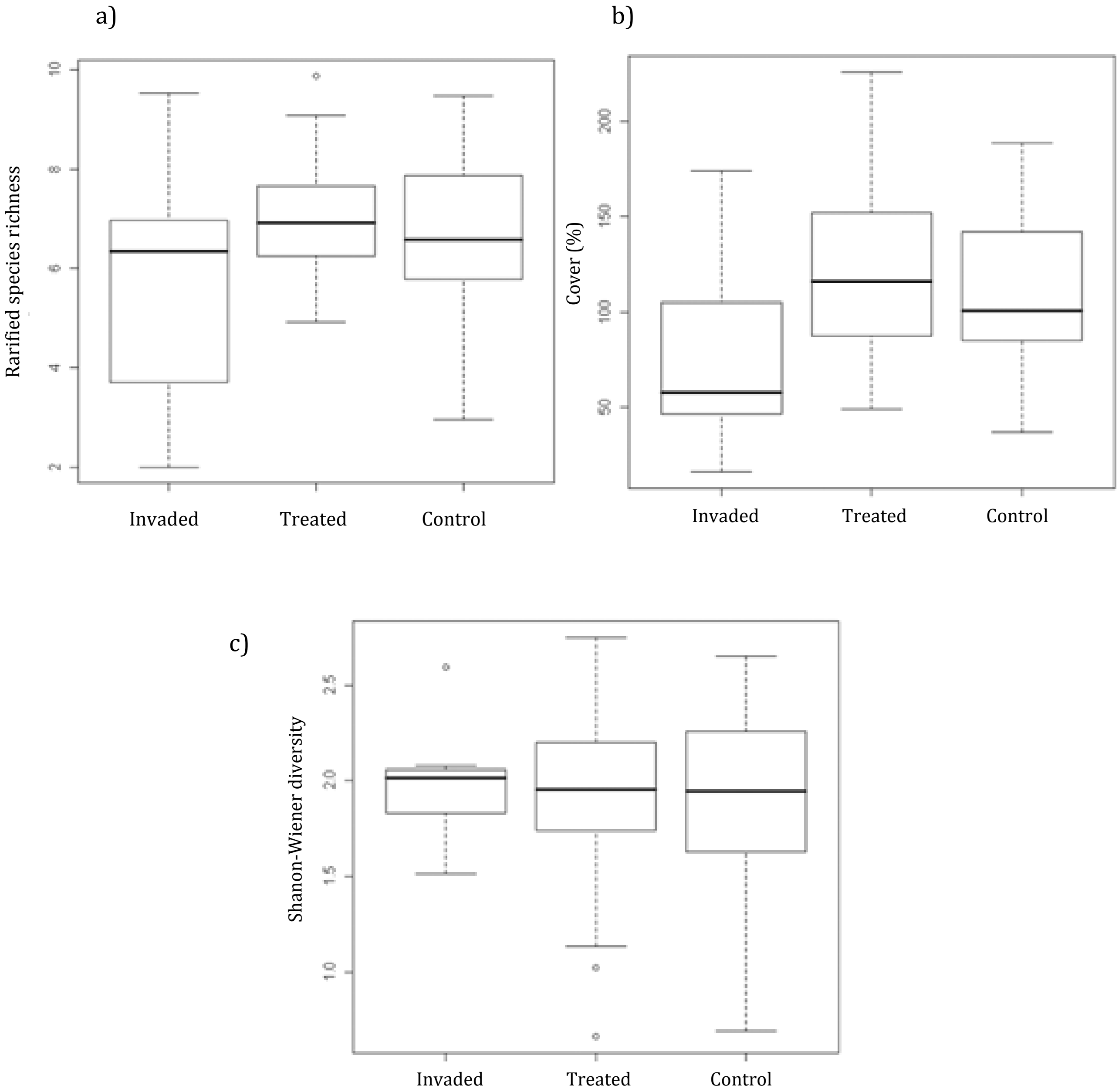


**Figure 5.3** Boxplots of native plant response to *Carpobrotus* removal in invaded, control, and treated plots in six coastal sites in Andalucía, Spain, measured by a) rarefied species richness; b) total native species cover; and c) Shannon-Wiener species diversity. The box itself contains 50% of the data (75th percentile indicated by the upper edge, the median by the center line, and the 25th percentile by the lower edge), with outliers as open circles and maximum/minimum value at the terminus of the vertical line.

### *Native species composition*

Despite the lack of changes in total native species cover and diversity, there were some changes in species composition. The Sorensen Similarity Index between control and treated plots was, on average, 0.77 ± 0.034, which provides additional support for the idea that regeneration after *Carpobrotus* removal results in coastal dune communities similar to reference native communities.

In Figure 5.4, percent cover of the different Raunkier functional groups have been compared between treated and control plots (Figure 5.4a) and between treated and invaded plots (Figure 5.4b). Only two of the five functional groups responded significantly to *Carpobrotus* removal. Cover of therophytes was significantly greater in treated plots than in control and invaded plots. This observed increase in annual plants suggests that the coastal dunes that were treated are in an early successional stage. Other studies have shown responses of annual plants following removal of invasive species (McCarthy 1997, Carlson & Gorchov 2004, Crimmins & McPherson 2008), which increases light, soil temperature, and resource availability, favoring the germination of species in the seed bank, such as annuals (D'Antonio & Meyerson 2002). However, cover of chamephytes (excluding *Carpobrotus*) was lower in treated plots than in control plots, and no significant differences were found between treated and invaded plots (Figure 5.4a, also see online appendix at uwpress.wisc.edu/journals/journals/er_suppl.html). This can be explained by the fact that chamephytes grow more slowly than therophytes and need more time to reestablish. As no significant differences in other functional groups were found between treated and control plots, we expect that natural community dynamics will lead them to become more mature communities with a more homogeneous relative cover of different life forms.

Without taking into account graminoids, a total of 107 species were found in our plots. Of these, 63 were never found in invaded plots, 27 of which appeared only in treated plots and 11 only in control plots. Another 43 species out of the 107 were not present in control plots, and only 11 species were never found in treated plots, of which 7 were present only in control plots.

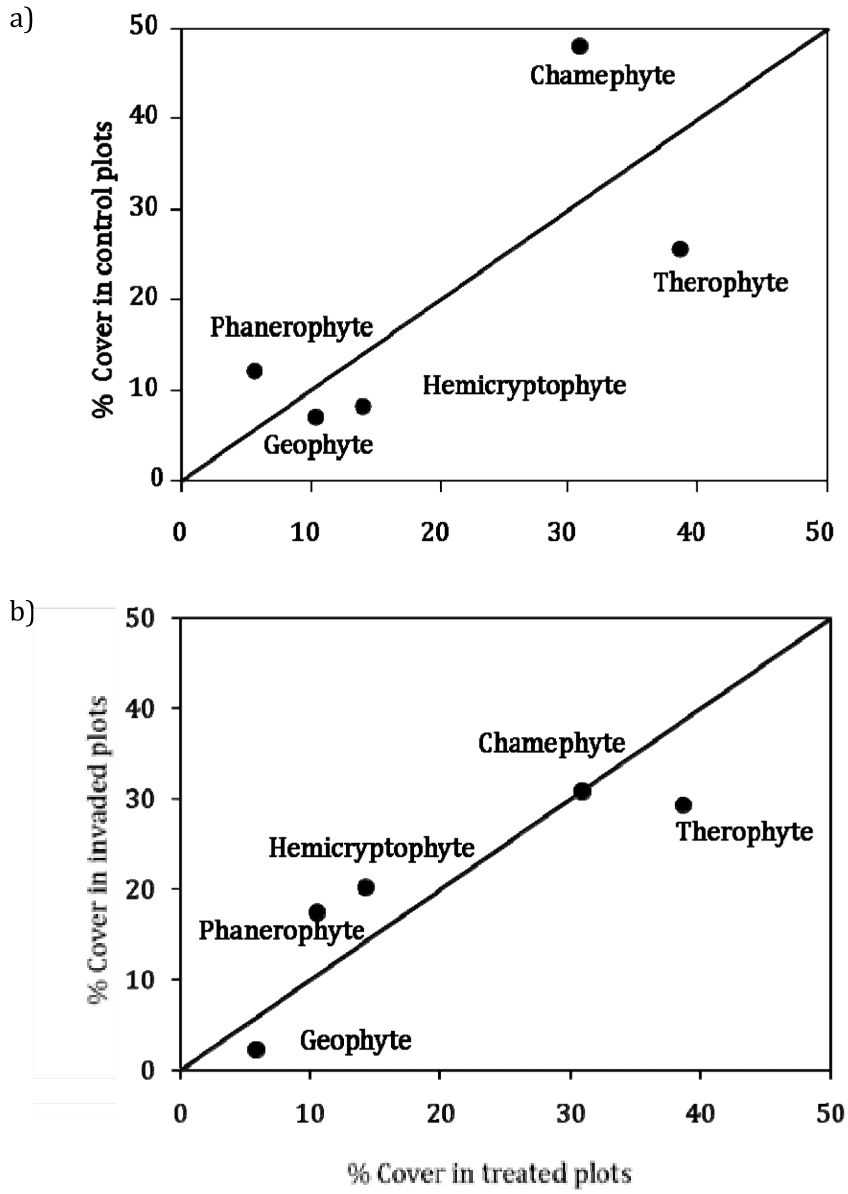


**Figure 5.4** Raunkier life-form cover as a function of treated plots for a) control and b) invaded plots. Proximity to the line of unity indicates lack of difference from the treated plots for the life-form group.

The relative cover of particular species differed between treatments (Figure 5.5). For example, water medick (*Medicago littoralis*), *Cyperus capitatus*, buckhorn plantain (*Plantago coronopus*), and whitebuttons (*Anacyclus clavatus*) were poorly represented in control plots and appeared very frequently in treated plots (Figure 5.5a).

On the contrary, creta trefoil (*Lotus creticus*), curry plant (*Helichrysum italicum*), *Helichrysum stoechas*, and *Rumex tingitanus* have higher relative cover in the control than in the treated plots (Figure 5.5a). In fact, neither *Helichrysum* was ever found in invaded plots, probably because they are associated with stable and undisturbed sites. There were also differences between invaded and treated plots. For instance, *Malcolmia littorea*, Geraldton carnation weed (*Euphorbia terracina*), and Engels gras (*Armeria pungens*) were more represented in invaded than treated plots, while *Cyperus capitatus* was as abundant in invaded as in treated plots (Figure 5.5b).

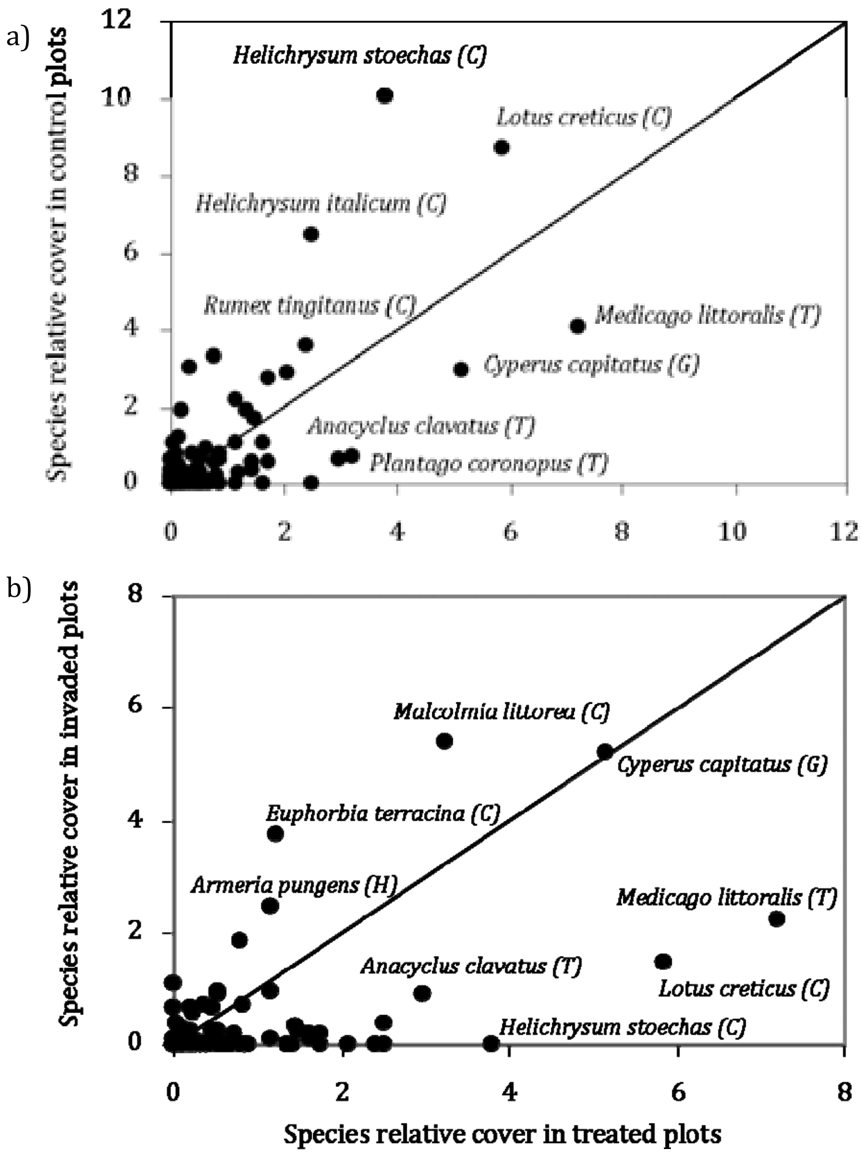


**Figure 5.5** Individual native species relative cover as a function of treated plots for a) control and b) invaded plots. Raunkiaer life-form category is indicated in parentheses: T = Therophyte, C = Chamephyte, G = Geophyte, H = Hemicryptophyte.

Only four of the 107 species were non-native. These were American century plant (*Agave americana*), Cape weed (*Arctotheca calendula*), salt heliotrope (*Heliotropium curassavicum*), and Bermuda buttercup (*Oxalis pes-caprae*). No significant differences were found between the total cover of non-native species in treated (4.78 ± 3.32), control (4.08 ± 2.25), and invaded (8 ± 6.33) plots (Kruskal-Wallis $H$ = 0.95, df = 2, $P$ = 0.620). Many studies have documented an increase in undesirable invasive species following disturbances (Burke & Grime 1996, Pickart et al. 1998, Zavaleta et al. 2001, Mason & French 2007, Crimmins & McPherson 2008). Such secondary invasions following control efforts can be problematic for ecological restoration (Hartman & McCarthy 2004, Hulme & Bremner 2006). The lack of such findings in our short-term study is, therefore, encouraging from a community management perspective.

## Conclusions and management implications

When eradicating *Carpobrotus* it is important to remove any remnants, as any remains left in place soon become an active focus of regeneration (Fraga et al. 2006), which was demonstrated by the very low densities of large recruits, probably resprouts, one year after pulling (Figure 5.2). Our research revealed that hand-pulling greatly reduced *Carpobrotus*; however, successful control will likely require perseverance and a commitment to long-term planning, implementation, and monitoring (Pickart et al. 1998, Manchester & Bullock 2000). Moreover, regional eradication would be needed in order to prevent new invasions from neighbouring populations (Pickart & Sawyer 1998).

Our findings suggest that native species could easily establish after *Carpobrotus* removal, particularly annual plants. However, with just one year of growth, these species are not able to occupy all bare ground (Díaz et al. 2003). In addition, comparisons between treated and control plots showed that management has resulted in coastal dunes with vegetation similar to reference native communities.

Some studies suggest that native species recovery after alien species removal requires several years. This could also be true for *Carpobrotus* and, therefore, our findings for these communities should not yet be regarded as definitive, since the managed sites are still in an early successional stage. Although repeated sampling is necessary to determine whether any observed pattern of community response is transient or stable (Sax & Brown

2000), these findings can be applied to achieve cost-effective removal strategies that accomplish overall restoration goals.

Overall, our results revealed that removal of *Carpobrotus* is not facilitating invasion by non-natives and that recovery of native species is high. This suggests that if seeds of native species are present, natural reestablishment is possible. Natural regeneration would be cheaper in these coastal dune communities than seeding after *Carpobrotus* removal. Thus, although planting desired native species is a potential scheme to facilitate the native recovery of a community, it is an expensive method and we do not consider it necessary in our case.

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# Chapter 6

# General discussion

## Alien plant management in Spain

Since the publication of Elton's (1958) classic *The Ecology of Invasions by Animals and Plants*, invasion biology has grown enormously, mostly in the last decade (Pyšek et al. 2006, Richardson & Pyšek 2008). At the moment, the scope of contemporary invasion biology is inevitably broad, ranging from essentially theoretical studies to practical recommendations of how to manage particular species and ecosystems (Pyšek & Richardson 2010). However, "management" is not yet a prevalent keyword topic in research on biological invasions (Pyšek et al. 2006).

Nonetheless, now that the increasing impacts and costs of invasive alien species are being recognized management of invasive species has become a high priority and an important challenge for land and protected areas managers (Bradley et al. 2010). Therefore, transfer of academic research results to management is highly needed.

**Chapter 2** sought to review the management measures for alien plants undertaken so far in Spain, to understand more fully the limitations and opportunities for effective management, and to illustrate how the scientific information percolates through management decisions (Hulme 2003, Bardsley & Edwards-Jones 2007, Daehler 2008, García-Llorente et al. 2008). Most of the time, managers have to deal with limited resources, which in turn require that choices must be made regarding where best to focus management efforts, which alien species to prioritize for management and which kind of management to use (Westman 1990, Hulme 2006, Sheley et al. 2010, Januchowski-Hartley et al. 2011).

According to questionnaire respondents, in Spain there are 193 alien plant species identified as noxious **(Chapter 2)**. However, only a little more than half of these species (109) are the subject of management actions. The answer to questionnaires also showed that managers in Spain are aware of the risks posed by biological invasions, which ranked similarly to any other global change environmental problems. These results are encouraging given that biological invasions have not been perceived as a problem by Spanish public authorities until very recently, or included in the environmental agenda (Capdevila-Argüelles et al. 2006). In Spain, management of plant invasions primarily focus on mechanical or chemical control measures at the local scale and, mostly with short-term goals in mind. Moreover, environmental managers differed in their views regarding how best to manage invasions. This probably reflects a lack of guidance and limited resources that perhaps results in less effective management (Westman 1990, Bardsley & Edward-Jones 2006). Our results demonstrated that there is clearly a lack of prevention measures and supervision of the longer term success of control actions. These two management strategies and the application of ecological knowledge to the management of biological invasions might lead to more cost-effective and viable strategies (Hulme 2006, Lodge et al. 2006, Smith et al. 2006). An ecosystem perspective of invasion management that integrates both the drivers of invasions as well as the target species control is also highly required (Hulme 2006).

Regarding the criteria used to prioritize invasive species management, **Chapter 2** illustrates that environmental managers clearly set priorities for management within their area of responsibility irrespective on the status of the species at a national scale. Rather, decisions are made based on local perceptions of abundance, distribution and perceived impact. This supports the idea that national priorities have to be based on an integrative risk analysis that combines information on both local and regional aspects (Hulme et al. 2008b). However, in the absence of guidelines for prioritization and the accompanying ecological knowledge to implement them, natural areas managers have to resort to more pragmatic and less scientific criteria for implementing policy. Sometimes the easiest plant species to control are given the most attention, regardless of the relative ecological damage they pose and the feasibility of their control (Westman 1990).

In conclusion, **Chapter 2** has contributed to improve the interaction between researchers and managers, and can be useful to inform further management decisions and to illustrate on effective management of invasive plant species in order to avoid future impacts. However, there still remains a pressing need to raise awareness about the impacts of alien species amongst both the general public, environmental managers and policy makers (Daehler 2008, Vanderhoeven et al. 2011). We also concur with Hulme (2003) who brought to light the lack of effective translation of academic research results of many areas of invasion ecology to management. More collaboration between academic research and environmental managers is required in order to achieve an efficient management of alien plants and to protect environmental integrity and native species diversity. Information about which species prioritize for control, which areas are most affected by alien plant species, and which are the most effective management methods for each species, is required in order to improve management success.

## Prevention

Prevention actions are recognized as more cost-effective and environmentally desirable than eradication or control measures (Pyšek & Richardson 2010). However, prevention of plant invasions is not common in Spain (**Chapter 2**). The rapidly increasing number of alien species and their tremendous costs to the environment and to the society (Hulme 2009, Kettunen et al. 2009) urgently require decision-making tools for the prevention of plant invasions and the regulation of plant trade in order to diminish the risks of new introductions. Risk assessment schemes are valuable tools to diminish the risk of invasion and to concentrate resources on preventing from entering pathways those species with higher risk of invasion (Pyšek & Richardson 2010).

Using two different risk assessments schemes, we have identified 80 potential invasive species not introduced yet to Spain, and that are already considered invasive in neighboring countries and other Mediterranean regions **(Chapter 3)**. These species could cause relevant economic and environmental impacts. The fact that the two risk schemes ranked species similarly increased the probability of having identified invasive plants correctly. Most plants identified as potentially invasive are ornamental species used in gardening, one of the major entry pathways of alien plants (Pyšek et al. 2002, Dehnen-Schmutz et al. 2007, Lambdon et al. 2008c). The species with the highest score in both tests and therefore with the highest risk to become invasive in Spain if introduced was *Chromolaena odorata* (L.) R.M.King & H.Rob. (Fam. Asteraceae). The other species with high scores also include five aquatic species: *Cabomba caroliniana* A. Gray (Fam. Cabombaceae*)*, *Hydrocotyle ranunculoides* L. f. (Fam. Apiaceae)*, Salvinia molesta* D. Mitch (Fam. Salviniaceae), *Ludwigia peploides* (Kunth) P.H. Raven (Fam. Onagraceae), *Alternanthera philoxeroides* (Mart.) Griseb. (Fam. *Amaranthaceae).* Aquatic specie*s* have great capacity of invasion, devastating impacts, not only environmental but also economic, and are complicated and costly to control (OTA 1993, Pimentel et al. 2005).

We recommend that the species with the highest scores in the risk assessments should be prohibited or kept out of trade. Besides, pathways in which species are specifically marketed such as aquarium trade, gardening and seed purchase should be examined and supervised. The high cost-benefit ratio associated with invasive species risk assessments (i.e., cost of allowing an invasion vs. benefit of allowing introduction of a presumed non-invasive) strongly recommends to use a very conservative approach. It is also important to take into account that alien plants can be invasive in one habitat of their introduced range but not in another, depending on particular habitat properties such as disturbance regime, microclimate/soil properties or propagule pressure (Thompson et al. 1995, Chytrý et al. 2008). As a consequence, although policies would need to be implemented at the national level, regional risk analyses within Spain are also necessary.

## Post-removal monitoring

Maintenance and restoration of ecosystems, where a particular invasive plant have been removed, and the subsequent monitoring have a significant influence on the success of management measures (Manchester & Bullock 2000, Hulme 2006, Smith et al. 2006).

Despite the importance of these measures, **Chapter 2** showed that in Spain they are often forgotten because control programs are usually designed with short-term goals in mind, mainly due to lack of sufficient economic resources or lack of guidance (Hobbs & Humphries 1995). Many times, simply eliminating a particular alien species may not lead to restore the original community. Managed sites can often be colonized by other alien species (Manchester & Bullock 2000, Simberloff 2003, Hulme & Bremner 2006, Loo et al. 2009). Thus, management programs should be designed to account for changes in native communities after invasion and after alien species removal (Zavaleta et al. 2001, SER 2002, Suding et al. 2004, Hulme 2006).

**Chapter 4** has quantified globally the change in native plant species richness and abundance, with plant invasion and after alien plant removal. The meta-analysis presented in **Chapter 4** revealed an overall significant decline of native species richness and abundance after invasion. Besides, it also provides evidence that after removal, native vegetation has the potential to recover to a pre-invasion target state. These results confirm the generalized negative impacts exerted by alien plants worldwide (Gaertner et al. 2009, Vilà et al. 2010) and corroborate that recovery to pre-existing conditions is possible (Hulme 2006). However, side-effects of the removal measures such us proliferation of other invasive species or changes of soil conditions should be also assessed in order to confirm our results (Álvarez & Cushman 2002, Ogden & Rejmánek 2005, Wolfe & Klironomos 2005, Hulme & Bremner 2006, Malcolm et al. 2008, Truscott et al. 2008, Loo et al. 2009).

The conceptual framework presented in **Chapter 4** highlights that observational and experimental approaches are needed to compare invaded, non-invaded and removal sites in order to quantify invaders' impacts and native plant recovery after alien species removal (See Figure 4.1 in **Chapter 4**). The review reveals that despite the complementarily between these comparisons they have been rarely used in concert. Many times reference sites are scarcely employed to assess native species recovery after removal because alien species are often so widespread by the time they are noticed that comparable non-invaded sites may be extremely rare or non-existing.

Finally, in **Chapter 5,** based on the conceptual framework presented in **Chapter 4,** we evaluated, as a case study, the efficacy of the manual removal of *Carpobrotus sp.* in coastal Andalucía, and the recovery of the native vegetation after this removal. The results verified the general trends found in **Chapter 4**. *Carpobrotus* reduces local species richness. The study also revealed that hand-pulling greatly reduced *Carpobrotus*; however, successful control will likely require perseverance and a commitment to long-term planning, implementation, and monitoring (Manchester & Bullock 2000, Simberloff 2009). Regarding the recovery of the native vegetation after removal, the study show that it is feasible but mainly annual native plants increase in cover after removal.

Comparisons between treated and control reference plots showed that management has resulted in coastal dunes with a vegetation similar to reference native communities. These results are in concordance with the general trend encountered in **Chapter 4**. Finally, we assessed possible side-effects of *Carpobrotus* removal. We found that the removal of *Carpobrotus* is not facilitating invasion by other alien species. Thus, since recovery of native species is high and new alien species are not colonizing, we can conclude that natural reestablishment is possible and would be cheaper in these coastal dune communities than seeding with native species. Nonetheless, we have to bear in mind that this is a particular case study. In other cases, planting desired native species in order to restore the invaded ecosystem can be a potential scheme to facilitate native community recovery and should be assessed (Hall & Hastings 2007).

**Chapter 4** and **5** provide evidence that post-removal monitoring is extremely helpful because it allows documenting the success of control and provides the opportunity to restrain negative side-effects before they become severe (Zavaleta et al. 2001, Simberloff 2009). Moreover, the evaluation of the native vegetation recovery is essential to determine whether further intervention, through active restoration techniques, is necessary. In fact, using removal methods without thoroughly testing their effectiveness and non-target effects can lead to routine implementation of inappropriate and costly techniques. Thus, we think that the conceptual framework presented in this thesis (see Figure 4.1 in **Chapter 4**) should be more frequently used because it may provide guidance for managers and may optimize management success.

# Conclusions

# (in Catalan)

# Conclusions

Tenint en compte que les mesures de control i eradicació d'espècies exòtiques són extremadament costoses tant en recursos humans i tècnics com econòmics (Pimentel et al. 2005, Andreu et al. 2009, Vilà et al 2010) és molt important, sempre que sigui possible, prevenir l'entrada i l'establiment d'espècies potencialment invasores, així com garantir l'eficàcia de les mesures de gestió aplicades, en aquells casos en què la prevenció ja no és possible (Manchester & Bullock 2000, Hulme 2006). Aquesta tesi omple una mica el buit d'informació entre el món acadèmic i el món de la gestió, proporcionant informació als gestors ambientals per tal de millorar l'eficàcia de les mesures de gestió aplicades.

***Gestió de les plantes exòtiques a Espanya***

I. Les invasions biològiques són considerades pels gestors ambientals entrevistats un problema de prioritat mitjana. A Espanya, un total de 109 espècies exòtiques problemàtiques han estat o estan sent gestionades. Els principals criteris que utilitzen els gestors a l'hora de prioritzar la gestió d'aquestes espècies són l'abundància local, la distribució àmplia i la percepció local d'impactes importants.

II. La gestió de les plantes exòtiques a Espanya s'ha centrat principalment, fins ara, en l'aplicació de mesures de control mecànic o químic dutes a terme a escala local i dissenyades, normalment, amb objectius a curt termini. Existeix, per tant, una manca important de mesures de prevenció, de mesures de seguiment a llarg termini i de pautes que facilitin la priorització de la gestió d'espècies exòtiques.

***Prevenció***

III. Mitjançant l'ús d'esquemes d'avaluació de riscos, s'ha elaborat una llista preliminar de 80 espècies potencialment invasores per Espanya, sent la família Leguminosae la més representada, i la jardineria la via d'entrada més comuna d'aquestes espècies. Les espècies amb les puntuacions més altes i, per tant, amb major risc d'esdevenir invasores, han resultat ser principalment plantes aquàtiques, i s'hauria de prohibir la seva entrada i el seu comerç. *Chromolaena odorata* (Asteraceae) és l'espècie que ha obtingut les puntuacions més altes i per tant podria ser considerada l'espècie amb major risc d'esdevenir invasora a Espanya, en cas que s'introduís al medi natural.

***Seguiment de les mesures de control***

IV. Aquesta tesi presenta un marc conceptual, basat en dades observacionals i experimentals, que compara llocs envaïts, llocs de referència no envaïts i llocs on una determinada espècie invasora ha estat eliminada. Aquestes comparacions permeten quantificar el impacte de l'espècie invasora en qüestió i supervisar la recuperació de les plantes natives després de la seva eliminació. En estudis científics revisats en aquesta tesi, els mètodes observacionals i experimentals no solen ser utilitzats de manera combinada, i els llocs de referència (no envaïts i no gestionats) són poc emprats per avaluar la recuperació d'espècies natives després de l'eliminació. Per tant, creiem que l'ús de l'enfocament comparatiu que es presenta en aquesta tesi hauria de ser més generalitzat. Es tracta d'una eina útil per determinar no només l'eficàcia de les mesures de control i/o eradicació i la recuperació de l'ecosistema natiu sinó que, a més, permet avaluar els possibles efectes secundaris de les tècniques d'eliminació, així com la necessitat de dur a terme mesures de restauració específiques a posteriori.

V. La revisió bibliogràfica global elaborada demostra que les plantes invasores són responsables d'una disminució local en la riquesa i l'abundància d'espècies natives. Això ha estat corroborat pel cas d'estudi sobre l'eliminació de *Carpobrotus* (Aizoaceae) a les dunes costaneres d'Andalusia, on aquesta espècie sembla també reduir significativament la riquesa d'espècies natives en les comunitats que envaeix.

VI. La revisió bibliogràfica global també indica que, en general, després de l'eliminació de les plantes invasores, la vegetació nativa té el potencial de recuperar-se fins arribar als nivells desitjats previs a la invasió. La mateixa tendència s'ha trobat per *Carpobrotus.* Donat que l'eliminació de *Carpobrotus* no sembla estar facilitant la proliferació d'altres espècies exòtiques i la recuperació de les espècies natives sembla ser elevada, podem concloure que en el cas de *Carpobrotus*, el restabliment i la recuperació de la vegetació nativa després de la seva eliminació són possibles.

# Conclusions

# (in English)

# Conclusions

Given that management measures are extremely costly in terms of human, technical and economic resources (Pimentel et al. 2005, Andreu et al. 2009, Vilà et al 2010) it is really important to, when possible, prevent potential invasive species as well as to guarantee management effectiveness in those cases where prevention is no longer a solution (Manchester & Bullock 2000, Hulme 2006). This thesis provides clear direction for bridging the current gap between the availability in information on alien plant species and the need for environmental managers to successfully prevent and control invasive species.

## *Alien plant management in Spain*

I. Biological invasions are considered by Spanish environmental managers as a medium priority problem. A total of 109 noxious alien species are being managed. The main criteria guiding managers to prioritize the allocation of resources when dealing with invasive plants were the local perception of being highly abundant, widely distributed and causing an impact.

II. Management of alien plants in Spain has mainly focused, so far, in the application of mechanical or chemical control measures at a local scale and mostly with short-term goals in mind. There is clearly a lack of preventive measures, absence of long-term monitoring of control actions and few guidelines for prioritization.

## *Prevention*

III. By using risk assessment schemes, we have identified a preliminary list of 80 potential invasive species, being Leguminosae the most represented family, and gardening the most common pathway of introduction. The species with the highest scores, were mainly aquatic plants, and should be prohibited or kept out of trade. *Chromolaena odorata* (Asteraceae) obtained the highest scores and therefore it might be the species with the highest risk to become invasive in Spain if introduced.

## *Post-removal monitoring*

IV. This thesis has presented a conceptual framework based on observational and experimental data that compares invaded, non-invaded and removal sites to quantify invaders' impacts and to monitor native plant recovery after their removal. In scientific studies, observational and experimental approaches are rarely used in concert, and reference sites are scarcely employed to assess native species recovery after removal. Thus, we believe that the comparative approach presented in this thesis should be more frequently used. Besides, it may also be useful to determine possible side-effects of removal techniques and whether further intervention is necessary.

V. Our global literature review has demonstrated that invasion by alien plants is responsible for a local decline in native species richness and abundance. This has been corroborated by monitoring *Carpobrotus* (Aizoaceae) invaded sites in coastal dunes of Andalucía, where this species has significantly decreased species richness by replacing native species in the communities it invades.

VI. Our global literature review has also indicated that, in general, after alien plant removal, the native vegetation has the potential to recover to a pre-invasion state. The same trend has been found for *Carpobrotus*. Since removal of *Carpobrotus* is not facilitating invasion by other alien species and recovery of native species seems to be high we have concluded that in this case study, a reestablishment of the native vegetation is possible.

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# Appendices

# Appendix I

Questionnaire on alien plants sent to environmental managers (**Chapter 2)**.

**Part 1: General questions on perception and identity of alien species of concern**

1. Which priority would you assign to the problem of biological invasions in relation to the other environmental problems of your area of responsibility?
    i. High priority
    ii. Medium priority
    iii. Low priority
2. Could you assign a number from 1 to 5 to the following environmental problems according to their priority or importance (1= low importance, 2= moderate importance, 3= important, 4= high importance, 5= extremely important)?

| | | | | | |
|---|---|---|---|---|---|
| Natural habitat loss | 1 | 2 | 3 | 4 | 5 |
| Habitat fragmentation | 1 | 2 | 3 | 4 | 5 |
| Wildfire | 1 | 2 | 3 | 4 | 5 |
| Biological invasions | 1 | 2 | 3 | 4 | 5 |
| Climate change | 1 | 2 | 3 | 4 | 5 |
| Pollution | 1 | 2 | 3 | 4 | 5 |
| Urbanization | 1 | 2 | 3 | 4 | 5 |
| Land use change | 1 | 2 | 3 | 4 | 5 |
| Other (specify which ones) | 1 | 2 | 3 | 4 | 5 |

3. Which priority would you assign to the following management strategies against invasions (ranking from 1 to 4; 1= low priority and 4= maximum priority)?
    i. Legislation reinforcement
    ii. Education and outreach
    iii. Entry prevention
    iv. Direct population control
2. Which are the main limitations or difficulties for an effective management of alien species in your area of responsibility?
3. Which alien species are causing problems (i.e., noxious alien plants) in your area of responsibility?

**Part 2: Questions for each alien species of concern**

**SPECIES 1:** *Species name*

1. Which kind of impacts is it causing?
    i. Ecological
    ii. Economic
    iii. Social

iv. Human health

2. Could you specify the impact type caused by this species?
3. Could you specify the magnitude of the impact caused by this species?
   i. High
   ii. Intermediate
   iii. Low
4. Could you mention any direct impact of this plant, which associated costs are easily quantified (i.e., infrastructure damage)? (*)
5. Is there any management strategy over this alien species in your area of responsibility? Yes/No
6. Which kinds of management activities do you carry out?
   i. Prevention - Outreach
      1. Regional legislation
      2. Education and information activities
      3. other (specify)
   ii. Eradication
   iii. Containment or population control
   iv. Restoration (i.e., habitat improvement, reforestation with native species)
7. How long have the management strategies been functioning?
8. Which method has been used to control or eradicate the alien species?
   i. Physical (mechanical, manual, etc.)
   ii. Chemical
   iii. Physical + Chemical
   iv. Biological control
9. With which frequency have the treatments been carried out?
10. Are the treatments carried out by technicians or volunteers?
11. Do you monitor the success of the management measures over the time?
12. How often do you monitor the state of the invasion?
13. Have you carried out a restoration of the locations previously invaded by the alien plant?
14. Could you estimate the total economic cost of the management measures?
    i. Prevention costs (three-page leaflets, workshops, conferences, etc.) = €.
    ii. Eradication or control costs (herbicides, salary, material, machinery, etc.)= €.
    iii. Habitat Restoration costs (native species plantation)= €.
15. Could you indicate which has been the result of the management measures?
    i. The species has been eliminated
    ii. The species has decreased considerably
    iii. The species has decreased very little
    iv. The species has not decreased
    v. The species continues to expand
16. Do you think that the management strategies have been successful?
    i. Very successful
    ii. Moderately successful
    iii. Not very successful
    iv. No successful at all

# Appendix II

List and status of the most noxious species according to respondents and the number of Autonomous Communities (ACs) and protected areas where noxious and managed (**Chapter 2**). An indication of their presence according to the Atlas of Alien Plants of Spain (Sanz-Elorza *et al.* 2004) is given in the 4th column. Status: I = invasive, N = naturalized, C = casual.

| Species (Family) | Status | Nº of ACs where noxious | Nº of ACs where managed | Nº of ACs where present | Nº of protected areas where noxious | Nº of protected areas where managed |
|---|---|---|---|---|---|---|
| *Carpobrotus spp.* (Aizoaceae) | I | 11 | 8 | 10 | 21 | 13 |
| *Eucalyptus spp.* (Myrtaceae) | I | 11 | 8 | 13 | 14 | 13 |
| *Ailanthus altissima* (Simaroubaceae) | I | 11 | 6 | 12 | 10 | 4 |
| *Robinia pseudoacacia* (Fabaceae) | I | 11 | 3 | 17 | 2 | 1 |
| *Acacia spp.* (Fabaceae) | I | 9 | 7 | 12 | 9 | 8 |
| *Cortaderia selloana* (Poaceae) | I | 8 | 7 | 11 | 2 | 2 |
| *Agave americana* (Agavaceae) | I | 7 | 3 | 12 | 14 | 2 |
| *Arundo donax* ( Poaceae) | I | 7 | 1 | 15 | 1 | 0 |
| *Opuntia spp.* (Cactaceae) | I | 7 | 4 | 13 | 9 | 3 |
| *Oxalis pes-caprae* (Oxalidaceae) | I | 7 | 2 | 11 | 4 | 2 |
| *Senecio spp.* (Asteraceae) | I | 6 | 3 | 11 | 4 | 2 |
| *Arctotheca calendula* (Asteraceae) | I | 5 | 3 | 10 | 2 | 2 |
| *Ipomoea spp.* (Convolvulaceae) | I | 5 | 3 | 14 | 1 | 1 |
| *Myoporum spp.* (Myoporaceae) | N | 5 | 4 | 3 | 1 | 1 |
| *Nicotiana glauca* (Solanaceae) | I | 5 | 2 | 8 | 5 | 2 |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Oenothera glazioviana* (Onagraceae) | I | 5 | 3 | 14 | 0 | 0 |
| *Paspalum spp.* (Poaceae) | I | 5 | 0 | 17 | 1 | 0 |
| *Ricinus communis* (Euphorbiaceae) | I | 5 | 1 | 8 | 2 | 2 |
| *Tradescantia fluminensis* (Commelinaceae) | I | 5 | 3 | 8 | 2 | 2 |
| *Aptenia cordifolia* (Aizoaceae) | N | 4 | 1 | 7 | 2 | 0 |
| *Aster squamatus* (Asteraceae) | I | 4 | 0 | 16 | 1 | 1 |
| *Baccharis halimifolia* (Asteraceae) | I | 4 | 4 | 3 | 0 | 0 |
| *Conyza spp.* (Asteraceae) | I | 4 | 0 | 17 | 0 | 0 |
| *Eichhornia crassipes* (Pontederiaceae) | I | 4 | 4 | 3 | 1 | 1 |
| *Xanthium strumarium* (Asteraceae) | I | 4 | 2 | 13 | 1 | 1 |
| *Aloe spp.* (Liliaceae) | C | 3 | 1 | 4 | 1 | 0 |
| *Amaranthus spp.* (Amaranthaceae) | I | 3 | 0 | 17 | 0 | 0 |
| *Azolla filiculoides* (Azollaceae) | I | 3 | 2 | 8 | 2 | 1 |
| *Buddleja davidii* (Buddlejaceae) | I | 3 | 2 | 7 | 1 | 1 |
| *Fallopia japonica* (Polygonaceae) | I | 3 | 2 | 6 | 0 | 0 |
| *Lantana spp.* (Verbenaceae) | I | 3 | 0 | 5 | 0 | 0 |
| *Oenothera biennis* (Onagraceae) | I | 3 | 2 | 14 | 0 | 0 |
| *Pennisetum setaceum* (Poaceae) | I | 3 | 1 | 3 | 2 | 2 |
| *Tropaeolum majus* (Tropaeolaceae) | I | 3 | 1 | 10 | 0 | 0 |
| *Yucca spp.* (Agavaceae) | C | 3 | 3 | - | 3 | 2 |
| *Araujia sericifera* (Asclepiadaceae) | I | 2 | 0 | 8 | 0 | 0 |
| *Artemisia spp.* (Asteraceae) | I | 2 | 0 | 13 | 1 | 0 |
| *Datura stramonium* (Solanaceae) | I | 2 | 2 | 16 | 3 | 2 |
| *Disphyma crassifolium* (Aizoaceae) | N | 2 | 0 | 4 | 1 | 0 |
| *Egeria densa* (Hydrocharitaceae) | N | 2 | 1 | 2 | 1 | 0 |
| *Kalanchoe spp.* (Crassulaceae) | C | 2 | 1 | - | 0 | 0 |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Ludwigia spp.* (Onagraceae) | N | 2 | 2 | 2 | 2 | 0 |
| *Oenothera drummondii* (Onagraceae) | N | 2 | 1 | 2 | 1 | 1 |
| *Pittosporum tobira* (Pittosporaceae) | C | 2 | 1 | 1 | 0 | 0 |
| *Platanus hybrida* (Platanaceae) | N | 2 | 2 | 8 | 0 | 0 |
| *Solanum bonariense* (Solanaceae) | I | 2 | 0 | 9 | 0 | 0 |
| *Sorghum halepense* (Poaceae) | I | 2 | 0 | 15 | 0 | 0 |
| *Spartina patens* (Poaceae) | I | 2 | 0 | 9 | 0 | 0 |
| *Xanthium spinosum* (Asteraceae) | N | 2 | 0 | 16 | 0 | 0 |

## Appendix III

WRA and WG-WRA scores for 80 potential invasive species in Spain (**Chapter 3**).

| Alien plant | WRA score | WG-WRA score | Alien plant |
|---|---|---|---|
| *Chromolaena odorata* (L.) R.M.King & H.Rob. | 27 | 33 | *Alternanthera philoxeroides* (Mart.) Griseb. |
| *Cabomba caroliniana* A.Gray | 27 | 33 | *Chromolaena odorata* (L.) R.M.King & H.Rob. |
| *Hydrocotyle ranunculoides* L. f. | 25 | 32 | *Salvinia molesta* D.Mitch. |
| *Salvinia molesta* D.Mitch. | 23 | 32 | *Prosopis glandulosa* Torr |
| *Hydrilla verticillata* (L.f.) Royle | 23 | 32 | *Cortaderia jubata* (Lem.) Stapf |
| *Prosopis glandulosa* Torr | 22 | 32 | *Heracleum mantegazzianum* Sommier & Levier |
| *Cryptostegia grandiflora* R.Br. | 22 | 32 | *Tamarix ramosissima* Ledeb. |
| *Ludwigia peploides*(Kunth) P.H.Raven | 21 | 31 | *Acacia mearnsii* De Wild. |
| *Alternanthera philoxeroides* (Mart.) Griseb. | 21 | 31 | *Panicum maximum* Jacq. |
| *Nassella tenuissima* (Trin.) Barkworth | 20 | 31 | *Tamarix aphylla* (L.) Karst. |
| *Cortaderia jubata* (Lem.) Stapf | 20 | 30 | *Cabomba caroliniana* A.Gray |
| *Panicum maximum* Jacq. | 19 | 30 | *Ludwigia peploides*(Kunth) P.H.Raven |
| *Elodea nuttallii* (Planch.) H.St.John | 19 | 30 | *Nassella tenuissima* (Trin.) Barkworth |
| *Crassula helmsii* (Kirk) Cockayne | 19 | 30 | *Acacia nilotica* (L.) Delile |
| *Asparagus asparagoides* (L.) Druce | 19 | 29 | *Cryptostegia grandiflora* R.Br. |
| *Acacia mearnsii* De Wild. | 19 | 29 | *Asparagus asparagoides* (L.) Druce |
| *Opuntia aurantiaca* Lindl. | 18 | 29 | *Mimosa pigra* L. |
| *Mimosa pigra* L. | 18 | 29 | *Hedychium gardnerianum* Sheppard ex Ker Gawl. |
| *Lupinus arboreus* Sims | 18 | 29 | *Mikania micrantha* Kunth |
| *Lagarosiphon major* (Ridl.) Moss | 18 | 28 | *Lagarosiphon major* (Ridl.) Moss |
| *Heracleum mantegazzianum* Sommier & Levier | 18 | 28 | *Hydrocotyle ranunculoides* L. f. |

| | | | |
|---|---|---|---|
| *Lysichiton americanus* Hultén & H.St.John | 17 | 28 | *Elodea nuttallii* (Planch.) H.St.John |
| *Clidemia hirta* D.Don | 17 | 28 | *Lupinus arboreus* Sims |
| *Watsonia bulbillifera* Matthews & L. Bolus | 16 | 28 | *Pueraria lobata* ( Willd. ) Ohwi |
| *Tamarix ramosissima* Ledeb. | 16 | 27 | *Hydrilla verticillata* (L.f.) Royle |
| *Tamarix aphylla* (L.) Karst. | 16 | 27 | *Alhagi pseudalhagi* (M.Bieb.) Desv. |
| *Hedychium gardnerianum* Sheppard ex Ker Gawl. | 16 | 27 | *Ligustrum sinense* Lour. |
| *Gunnera tinctoria* (Molina) Mirb. | 16 | 27 | *Berberis thunbergii* DC. |
| *Cotoneaster franchetii* Boiss. | 16 | 27 | *Rosa rugosa* Thunb. |
| *Rubus ellipticus* Sm. | 15 | 26 | *Crassula helmsii* (Kirk) Cockayne |
| *Pueraria lobata* ( Willd. ) Ohwi | 15 | 26 | *Lysichiton americanus* Hultén & H.St.John |
| *Miscanthus sinensis* Anderss. | 15 | 26 | *Gunnera tinctoria* (Molina) Mirb. |
| *Miconia calvescens* DC. | 15 | 26 | *Rubus ellipticus* Sm. |
| *Epilobium ciliatum* Raf. | 15 | 26 | *Reynoutria sachalinensis* (F.Schmidt) Nakai |
| *Cereus martinii* Labour. | 15 | 25 | *Opuntia aurantiaca* Lindl. |
| *Acacia nilotica* (L.) Delile | 15 | 25 | *Dalbergia sissoo* Roxb. ex DC. |
| *Parthenium hysterophorus* L. | 14 | 25 | *Leptospermum laevigatum* F. Muell. |
| *Mikania micrantha* Kunth | 14 | 25 | *Passiflora subpeltata* Ortega |
| *Triadica sebifera* (L.) Small | 13 | 25 | *Reynoutria x bohemica* Chrtek & Chrtková |
| *Sesbania punicea* (Cav.) Benth. | 13 | 24 | *Clidemia hirta* D.Don |
| *Ligustrum sinense* Lour. | 13 | 24 | *Cotoneaster franchetii* Boiss. |
| *Dalbergia sissoo* Roxb. ex DC. | 13 | 24 | *Cereus martinii* Labour. |
| *Chorispora tenella* (Pall.) DC. | 13 | 24 | *Epilobium ciliatum* Raf. |
| *Celtis sinensis* Pers. | 13 | 24 | *Miconia calvescens* DC. |
| *Alhagi pseudalhagi* (M.Bieb.) Desv. | 13 | 24 | *Miscanthus sinensis* Anderss. |
| *Acroptilon repens* (L.) DC. | 13 | 24 | *Triadica sebifera* (L.) Small |
| *Psidium cattleianum* Sabine | 12 | 24 | *Coreopsis lanceolata* L. |
| *Passiflora subpeltata* Ortega | 12 | 24 | *Psidium cattleianum* Sabine |

| | | | |
|---|---|---|---|
| *Leptospermum laevigatum* F. Muell. | 12 | 24 | *Spathodea campanulata* P.Beauv. |
| *Heracleum sosnowskyi* Mandenova | 12 | 23 | *Watsonia bulbillifera* Matthews & L. Bolus |
| *Berberis thunbergii* DC. | 12 | 23 | *Celtis sinensis* Pers. |
| *Acacia paradoxa* DC. | 12 | 23 | *Heracleum sosnowskyi* Mandenova |
| *Spathodea campanulata* P.Beauv. | 11 | 23 | *Sesbania punicea* (Cav.) Benth. |
| *Reynoutria x bohemica* Chrtek & Chrtková | 11 | 23 | *Acacia paradoxa* DC. |
| *Reynoutria sachalinensis* (F.Schmidt) Nakai | 11 | 23 | *Cecropia peltata* L. |
| *Coreopsis lanceolata* L. | 11 | 23 | *Melaleuca quinquenervia* (Cav.) S.T.Blake |
| *Cinnamomum camphora* (L.) J. Presl | 11 | 23 | *Sphagneticola trilobata* (L.) Pruski |
| *Sphagneticola trilobata* (L.) Pruski | 10 | 23 | *Passiflora edulis* Sims |
| *Rosa rugosa* Thunb. | 10 | 22 | *Acroptilon repens* (L.) DC. |
| *Melaleuca quinquenervia* (Cav.) S.T.Blake | 10 | 22 | *Cinnamomum camphora* (L.) J. Presl |
| *Eugenia uniflora* L. | 10 | 22 | *Hakea salicifolia* (Vent.) B.L.Burtt |
| *Cinchona pubescens* Vahl | 10 | 22 | *Amelanchier spicata* (Lam.) K.Koch |
| *Cecropia peltata* L. | 10 | 22 | *Solidago nemoralis* Ait. |
| *Aristolochia elegans* Mast. | 10 | 21 | *Ardisia elliptica* Thunb. |
| *Ardisia elliptica* Thunb. | 10 | 21 | *Aristolochia elegans* Mast. |
| *Humulus scandens* (Lour.) Merr. | 9 | 21 | *Spartina anglica* C.E.Hubb. |
| *Dipogon lignosus* (L.) Verdc. | 9 | 20 | *Parthenium hysterophorus* L. |
| *Hakea salicifolia* (Vent.) B.L.Burtt | 8 | 20 | *Cinchona pubescens* Vahl |
| *Solidago nemoralis* Ait. | 7 | 20 | *Eugenia uniflora* L. |
| *Senna septemtrionalis* (Viv.) H.S.Irwin & Barneby | 7 | 20 | *Humulus scandens* (Lour.) Merr. |
| *Amelanchier spicata* (Lam.) K.Koch | 7 | 20 | *Echinocystis lobata* (Michx.) Torr. & A.Gray |
| | | 20 | *Hiptage benghalensis* (L.) Kurz |
| | | 20 | *Syzygium cumini* (L.) Skeels |
| | | 19 | *Solanum seaforthianum* Andrews |

| | | |
|---|---|---|
| | 18 | *Dipogon lignosus* (L.) Verdc. |
| | 18 | *Senna septemtrionalis* (Viv.) H.S.Irwin & Barneby |
| | 18 | *Bidens connata* Muhl. ex Willd. |
| | 18 | *Pinus elliottii* Engelm. |
| | 16 | *Rivina humilis* L. |
| | 14 | *Chorispora tenella* (Pall.) DC. |

## Appendix IV

References used to construct the dataset on native plant richness and abundance responses to alien plant invasion and removal **(Chapter 4).**

### A) Non-invaded vs. invaded comparisons

| Species | Life form | Invaded region | Biogeographical region | Invaded habitats | Variable measured | Reference |
|---|---|---|---|---|---|---|
| *Agave americana* | Shrub | Europe | Mediterranean | Coastal (sand dunes/rocky) | Abundance | Badano & Pugnaire 2004 |
| *Agropyron cristatum* | Perennial herb | North America | Temperate | Grassland | Abundance | Henderson & Naeth 2005 |
| *Agrostis stolonifera* | Perennial grass | Asia | Temperate | Riparian | Abundance | Gremmen et al. 1998 |
| *Ailanthus altissima* | Tree | Europe | Mediterranean | Riparian | Richness | Vilà et al. 2006 |
| *Asparagus asparagoides* | Shrub | Australia | Subtropical | Grassland | Abundance, richness | Turner et al. 2008 |
| *Bromus tectorum* | Annual grass | North America | Semiarid | Grassland | Abundance | Belnap & Phillips 2001 |
| *Bromus tectorum* | Annual grass | North America | Semiarid | Grassland | Abundance | Belnap et al. 2005 |
| *Bromus tectorum* | Annual grass | North America | Semiarid | Grassland | Abundance | Belnap et al. 2006 |
| *Bromus tectorum* | Annual grass | North America | Mediterranean | Grassland | Abundance | Blank 2008 |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Carpobrotus edulis* | Perennial herb | Europe | Mediterranean | Coastal (sand dunes/rocky) | Abundance, richness | Andreu et al. 2009 |
| *Carpobrotus spp.* | Perennial herb | Europe | Mediterranean | Coastal (sand dunes/rocky) | Richness | Vilà et al. 2006 |
| *Chromolaena odorata* | Perennial herb | Asia | Subtropical | Forest | Richness | Muralli & Setty 2001 |
| *Chrysanthemoides monilifera* ssp. *rotundata* | Shrub | Australia | Subtropical | Coastal (sand dunes/rocky) | Richness | Mason & French 2008 |
| *Chrysanthemoides monilifera* ssp. *rotundata* | Shrub | Australia | Subtropical | Coastal (sand dunes/rocky) | Richness | Mason et al. 2007. |
| *Cortaderia selloana* | Perennial grass | Europe | Mediterranean | Oldfields | Abundance, richness | Domenech & Vilà 2008 |
| *Cytisus scoparius* | Shrub | Australia | Temperate | Forest | Richness | Wearne & Morgan 2004 |
| *Delairea odorata* | Vine | North America | Mediterranean | Shrubland | Abundance, richness | Alvarez & Cushman 2002 |
| *Heracleum mantegazzianum* | Perennial herb | Europe | Temperate | Several habitats | Richness | Pyšek & Pyšek1995 |
| *Impatiens glandulifera* | Annual herb | Europe | Temperate | Riparian | Richness | Hejda & Pyšek 2006 |
| *Lantana camara* | Shrub | Asia | Subtropical | Forest | Richness | Muralli & Setty 2001 |
| *Lantana camara* | Shrub | Australia | Semi-arid | Forest | Abundance, richness | Gooden et al. 2009 |
| *Lonicera tatarica* | Shrub | North America | Temperate | Forest | Abundance, richness | Woods 1993 |
| *Lupinus polyphyllus* | Perennial herb | Europe | Temperate | Grassland | Abundance, richness | Valtonen et al. 2006 |
| *Lythrum salicaria* | Perennial herb | North America | Temperate | Grassland | Richness | Treberg & Husband 1999 |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Mesembryanthemum crystallinum* | Annual herb | Africa | Mediterranean | Coastal (sand dunes/rocky) | Abundance | Vivrette & Muller 1977 |
| *Mimulus guttatus* | Perennial herb | Europe | Oceanic | Grassland | Richness | Trusctott et al. 2008 |
| *Orbea variegata* | Perennial herb | Australia | Mediterranean | Shrubland | Abundance, richness | Dunbar & Facelli 1999 |
| *Oxalis pes-caprae* | Perennial herb | Europe | Mediterranean | Oldfields | Richness | Vilà et al. 2006 |
| *Phragmites australis* | Perennial grass | North America | Temperate | Grassland | Abundance, richness | Richburg et al. 2001 |
| *Pyracantha angustifolia* | Shrub | South America | Subtropical | Shrubland | Richness | Giantomasi et al. 2008 |
| *Spartina anglica* | Perennial grass | North America | Temperate | Wetland/marshland | Abundance | Hacker & Dethier 2006 |
| *Tradescantia fluminensis* | Perennial herb | Pacific islands | Subtropical | Forest | Abundance | Standish et al. 2001 |

**B) Removal vs. invaded comparisons**

| Species | Life form | Region invaded | Biogeographical region | Invaded habitats | Variable measured | Reference |
|---|---|---|---|---|---|---|
| Annual grasses from the genus *Bromus* | Annual grass | North America | Subtropical | Desert | Abundance | Brooks 2000 |
| *Bromus tectorum* | Annual grass | North America | Temperate | Shrubland | Abundance | Melgoza et al. 1990 |
| *Carpobrotus edulis* | Perennial herb | Europe | Mediterranean | Coastal (sand dunes/rocky) | Abundance, richness | Andreu et al. 2009 |
| *Chrysanthemoides monilifera* ssp. *rotundata* | Shrub | Australia | Semi-arid | Coastal (sand dunes/rocky) | Richness | Mason & French 2008 |
| *Delairea odorata* | Vine | North America | Mediterranean | Riparian, shrubland | Richness | Alvarez & Cushman 2002 |
| *Impatiens glandulifera* | Annual herb | Europe | Temperate | Riparian | Richness | Hejda & Pyšek 2006 |
| *Impatiens glandulifera* | Annual herb | Europe | Temperate | Riparian | Richness | Hulme & Bremner 2006 |
| *Lantana camara* | Shrub | Australia | Semi-arid | Forest | Abundance, richness | Gooden et al. 2009 |
| *Lespedeza cuneata* | Perennial herb | North America | Temperate | Grassland | Abundance | Brandon et al. 2004 |
| *Lupinus arboreus* | Shrub | North America | Mediterranean | Shrubland | Abundance | Pickart et al. 1998 |
| *Microstegium vimineum* | Annual grass | North America | Temperate | Forest | Abundance, richness | Oswalt et al. 2007 |
| *Microstegium vimineum* | Annual grass | North America | Temperate | Forest | Abundance | Flory 2010 |

| | | | | | | |
|---|---|---|---|---|---|---|
| *Microstegium vimineum* | Annual grass | North America | Temperate | Forest | Abundance, richness | Flory & Clay 2009 |
| *Mimulus guttatus* | Perennial herb | Europe | Oceanic | Grassland | Richness | Trusctott et al. 2008 |
| *Orbea variegata* | Perennial herb | Australia | Mediterranean | Shrubland | Abundance, richness | Dunbar & Facelli 1999 |
| *Tamarix* spp. | Tree | North America | Temperate | Forest | Abundance, richness | Harms & Hiebert 2006 |

**C) Non-invaded vs. removal comparisons**

| Species | Life form | Region invaded | Biogeographical region | Invaded habitats | Variable measured | Reference |
|---|---|---|---|---|---|---|
| *Carpobrotus edulis* | Perennial herb | Europe | Mediterranean | Coastal (sand dunes/rocky) | Abundance, richness | Andreu et al. 2009 |
| *Chrysanthemoides monilifera* ssp. *rotundata* | Shrub | Australia | Semi-arid | Coastal (sand dunes/rocky) | Richness | Mason & French 2008 |
| *Lantana camara* | Shrub | Australia | Semi-arid | Forest | Abundance, richness | Gooden et al. 2009 |
| *Mimulus guttatus* | Perennial herb | Europe | Oceanic | Grassland | Richness | Trusctott et al. 2008 |
| *Orbea variegata* | Perennial herb | Australia | Mediterranean | Shrubland | Richness | Dunbar & Facelli 1999 |
| *Pinus radiata* and *Hakea sericea* | Tree | Africa | Mediterranean | Forest | Abundance, richness | Holmes et al. 2000 |

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