

Private protected areas in Australia: current status and future directions

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Abstract

Despite the recognised importance of private land for biodiversity conservation, there has been little research into systems of private protected areas at a country-wide level. Here I look at definitions, legislation, ownership, management approaches and effectiveness, distribution and incentives provided to private protected areas in Australia. The term ‘private protected areas’, although increasingly used, still suffers from a lack of a clear and concise definition in Australia. Australian states and territories have legislation enabling the application of conservation covenants over private land; covenants being the primary mechanism to secure conservation intent on the title of the land in perpetuity. If considering all ‘in perpetuity’ conservation covenants under a dedicated program to be private protected areas and land owned by non-government organisations and managed for the purpose of biodiversity conservation, there were approximately 5,000 terrestrial properties that could be considered private protected areas in Australia covering 8,913,000 ha as at September 2013. This comprises almost 4,900 conservation covenants covering over 4,450,000 ha and approximately 140 properties owned by private land trusts covering approximately 4,594,120 ha. Most conservation covenanting programs now seek to complement the comprehensiveness, adequacy and representativeness of the public reserve system, either stating so explicitly or by aiming to protect the highest priority ecosystems on private land. There are a range of incentives offered for private land conservation and requirements of owners of private protected areas to report on their activities vary in Australia. However, there are a number of key policy challenges that need to be addressed if private protected areas are to achieve their full potential in Australia, including managing broad-scale ecosystem processes, protection and tenure reform, improved financial incentives, and access to emerging ecosystem service markets.

Keywords

National Reserve System, conservation covenants, private reserves, land trusts, legislation, ownership, incentives

Introduction

The commitment by most countries to expand the protected area estate in a representative and well-connected manner, as part of the Convention on Biological Diversity's Aichi Target 11, will require the inclusion of a range of protection mechanisms over a variety of tenures, including protected areas over private land (Woodley et al. 2012). Despite their potentially important role in biodiversity conservation, recognition of the role of private protected areas has suffered from sparse data, loose definitions and lack of integration into other protected area estates (Stolton et al. 2014). In a recent global review of private protected areas, Stolton et al. (2014) suggested Australia had a 'well developed' and 'vibrant' system of private protected areas (along with other countries such as Brazil, Chile, Colombia, Mexico, South Africa and the USA). Here, I look at the development of the private protected area estate in Australia, which has seen a dramatic growth in area and number of properties permanently protected for nature conservation, but has received little attention in the literature. Specifically I address the definitions, outline the legislation, ownership, management approaches and effectiveness, distribution and incentives provided to private protected areas on the Australian continent, highlight challenges and suggest future directions.

In Australia, the conservation of biodiversity on private land has been an important policy objective for the past few decades (e.g., Commonwealth of Australia 1996; Natural Resource Management Ministerial Council 2009, 2010). While there are multiple mechanisms used to achieve this, conservation covenants and land acquisition are the primary mechanisms used to protect natural assets on private land in the long-term (Fitzsimons and Wescott 2001; Figgis et al. 2005; Cowell and Williams 2006; Pasquini et al. 2011). A conservation covenant is a binding agreement (usually entered into on a voluntary basis) between a landowner and an authorised body to help the landowner protect and manage the environment on their property. There is a variety of conservation covenanting mechanisms with supporting programs that currently exist in Australia. Conservation covenanting programs vary across Australia, based on the jurisdiction and the legislation under which they are established. All of these are statutory mechanisms, with the covenants established through specific legislation. The programs have a variety of origins, the oldest being established in the late 1970s in Victoria (although the first 'wildlife refuge' was signed in the 1950s in New South Wales) and some more recent programs that have only been operating in the last few years.

The Australian National Reserve System is a national network of public, Indigenous and private protected areas over land and inland freshwater. Its focus is to secure long-term protection for samples of Australia's diverse ecosystems and the plants and animals they support. It is recognised that the National Reserve System

cannot be built solely on public lands and there is a significant role for Indigenous groups, local communities, private landholders and non-government organisations to play in establishing and managing protected areas to ensure the success of the National Reserve System. The Australian Government has played an important role in growing the private land trust sector in Australia over the past 20 years (land trusts being non-government organisations owning and managing land for conservation). Specifically, the provision of up to two-thirds of the purchase price for strategic land acquisitions through the National Reserve System program has seen land owned by this sector grow from thousands of hectares in the mid-1990s to millions of hectares today. It has also resulted in significantly increased involvement and investment from the philanthropic sector in the establishment of new private protected areas (Humann 2012; Taylor 2012; Taylor et al. 2014).

How is a private protected area defined in Australia?

The term ‘private protected areas’, although increasingly used, still suffers from a lack of a clear and concise definition in Australia. In this paper, land held for conservation by Indigenous people and groups while substantial in Australia (Rose 2012) are not considered ‘private’ for the purpose of protected area governance classifications. Rather they are considered to fall into the ‘Indigenous’ governance category of the IUCN’s protected area framework (Dudley 2008). The only *nationally agreed* definition of private protected area is that developed by the Natural Resource Management Ministerial Council (NRMMC) for *Australia’s Strategy for the National Reserve System 2009–2030* (NRMMC 2009). The Natural Resource Management Ministerial Council, which consisted of the Australian Commonwealth, state, territory and New Zealand government ministers responsible for primary industries, natural resources, environment and water policy, stated “A fundamental requirement of any area’s eligibility for inclusion within the National Reserve System is that it must meet the IUCN definition of a ‘protected area’ (Dudley 2008)” (NRMMC 2009, p. 42). The Natural Resource Management Ministerial Council (2009, p. 42) defined further ‘Standards for inclusion in the National Reserve System’ with three standards applying generally across all tenure types and a fourth (dealing with security) specific to different tenures (i.e. public, private, Indigenous) (Table 1).

The Natural Resource Management Ministerial Council (2009, p. 43) provides further definition of the term ‘legal or other effective means’ for the purposes of inclusion in the National Reserve System:

1. Legal means: Land is brought under control of an Act of Parliament, specialising in land conservation practices, and requires a Parliamentary process to extinguish the protected area or excise portions from it.
2. Other effective means: for contract, covenant, agreements or other legal instrument, the clauses must include provisions to cover:

Table 1. Standards for inclusion in the National Reserve System (source: Natural Resource Management Ministerial Council 2009).

Standards	Description
Valuable	<ul style="list-style-type: none"> • must enhance the comprehensiveness, adequacy and representativeness of the National Reserve System • must be established and managed for the primary purpose of protection and maintenance of biological diversity with associated ecosystem services and cultural value
Secure through legal or other effective means	<i>Public</i>
	<ul style="list-style-type: none"> • must be statutorily defined and resourced
	<i>Private</i>
Secure through legal or other effective means	<ul style="list-style-type: none"> • must be reserved in perpetuity • any change in management status must have Ministerial or statutory approval
	<i>Indigenous</i>
	<ul style="list-style-type: none"> • must have customary law protection with Traditional Owners holding a non-transferable interest in the land with a commitment to its long-term protective management • must be a commitment from Traditional Owners to discuss any changes with the Minister
Well-managed	<ul style="list-style-type: none"> • must be classified and managed in accordance with one or more IUCN management categories (I–VI) • must be adaptively managed to minimise loss of biodiversity values • effectiveness of management must be monitored and evaluated in a manner open to public scrutiny
Clearly defined	<ul style="list-style-type: none"> • the area must be able to be accurately identified on maps and on the ground

- long-term management – ideally this should be in perpetuity but, if this not possible, then the minimum should be at least 99 years;
- the agreement to remain in place unless both parties agree to its termination;
- a process to revoke the protected area or excise portions from it is defined; for National Reserve System areas created through contribution of public funding, this process should involve public input when practicable;
- the intent of the contract should, where applicable, be further reinforced through a perpetual covenant on the title of the land; and
- ‘well-tested’ legal or other means, including non-gazetted means, such as through recognised traditional rules under which Indigenous Protected Areas (community conserved areas) operate or the policies of established non-government organisations.

This definition largely reflects previous definitions of the Natural Resource Management Ministerial Council (2005) in its *Directions for the National Reserve System – A Partnership Approach* with the exception of the last point which is new to the ‘Strategy’. Fitzsimons (2006) provided a detailed analysis of how each private land conservation mechanism in the State of Victoria met the definition of private protected area (based on the Natural Resource Management Ministerial Council 2005 definition), however it does not appear that similar analyses have been carried out for other jurisdictions.

Nonetheless, conservation covenants, land purchased by non-government organisations through the National Reserve System Program, and less frequently, areas protected

by special legislation or under the National Parks legislation, are the main ‘types’ of private protected areas in Australia and this is the focus of the discussion below.

However, it should be recognised, that despite the definitions above, the term ‘private protected areas’ is often used more broadly for private land conservation mechanisms that include a legislative or contractual component (even if not in perpetuity) or generally for land owned by conservation land trusts or similar.

Legislation that addresses private protected areas in Australia

In Australia, as the environment was not listed as an item in the Australian constitution at Federation, state and territory governments are primarily responsible for environmental management and relevant legislation (Wescott 1991). This includes protected area legislation to enable the creation of public protected areas (typically ‘National Parks Acts’). The states and territories also have legislation enabling the application of conservation covenants over private land; covenants being the primary mechanism to secure conservation intent on the title of the land in perpetuity. Some states have more than one piece of legislation that enables conservation covenants, and the Australian Government also has a mechanism that allows covenants to be signed, although this is little used. The conservation covenanting programs and their respective legislation are presented in Table 2.

Where financial assistance has been given to non-government organisations to purchase land for conservation through the Australian Government’s National Reserve System program, protection takes two main forms. Firstly, there is a funding agreement between the Australian Government and non-government organisation which specifies the purpose of the property being for biodiversity conservation, the management activities to be undertaken and activities which are not appropriate. There is provision in many of these agreements for funding to be returned if provisions are not met. Critically there is a requirement in all contracts for a conservation covenant (or similar) to be signed between the non-government organisation with the relevant state/territory covenanting agency as soon as possible after purchase.

In South Australia, the government has proposed to amend the *National Parks and Wildlife Act 1972* to allow the establishment of National Parks and Conservation Parks on private freehold and leasehold lands (Leaman and Nicolson 2012). In this proposal the land owner would enter into an agreement with the Minister, the park would be declared and a notation would be included on the land title. Under this model, National Parks and Conservation Parks on private land would remain under the control and management of the landholder in accordance with a management plan prepared by the owner and approved by the Minister. However, the terminology met with resistance and as a result of the feedback, current thinking is to amend the proposal to maintain the underlying concept, but move away from the terms ‘National Park’ and ‘Conservation Park’. The term ‘Private Reserve’ seems to have broader acceptance and is being considered as an alternative (Leaman and Nicolson 2012).

Table 2. Covenanting programs in Australian jurisdictions and primary legislation.

Jurisdiction	Program	Legislation
Australian Government	Conservation Agreements †	<i>Environment Protection and Biodiversity Conservation Act 1999</i>
Western Australia	National Trust of Australia (WA) Covenanting Program	<i>National Trust of Australia (WA) Act 1964</i> and <i>Transfer of Land Act 1893</i>
Western Australia	Nature Conservation Covenant Program	<i>Conservation and Land Management Act 1984</i> and <i>Transfer of Land Act 1893</i>
Western Australia	Soil and Land conservation covenants	<i>Soil and Land Conservation Act 1945</i>
South Australia	South Australian Heritage Agreement Program	<i>Native Vegetation Act 1991</i>
Victoria	Trust for Nature (Victoria) conservation covenants	<i>Victorian Conservation Trust Act 1972</i>
Victoria	Land Management Co-operative Agreements	<i>Conservation, Forests and Lands Act 1987</i>
Tasmania	Private Property Conservation Program (Now includes sub programs of Protected Areas on Private Land (PAPL) and Non-Forest Vegetation Program)	<i>Nature Conservation Act 2002</i> and <i>Land Titles Act 1980</i>
New South Wales	Voluntary Conservation Agreements Program	<i>National Parks and Wildlife Act 1974</i>
New South Wales	Wildlife Refuges	<i>National Parks and Wildlife Act 1974</i>
New South Wales	Nature Conservation Trust covenants	<i>Nature Conservation Trust Act 2001</i>
New South Wales	NSW Registered Property Agreements Program	<i>Native Vegetation Act 2003</i>
Queensland	Queensland Nature Refuge program	<i>Nature Conservation Act 1992</i> and <i>Nature Conservation (Protected Areas) Regulations 1994</i>
Queensland	Voluntary conservation agreement programs operated by south-east Queensland councils, including Gold Coast, Sunshine Coast, Moreton Bay, Brisbane and Logan Local Governments	<i>Queensland Land Title Act 1994</i>
Northern Territory	Voluntary conservation covenant program	<i>Parks and Wildlife Commission Act 2004</i> and <i>Land Title Act 2007</i>

Notes: † Only a few Conservation Agreements signed under the *Environment Protection and Biodiversity Conservation Act* could be considered to be akin to a covenant – see <http://www.environment.gov.au/epbc/about/conservation-agreements.html#list>

Unlike most national parks in Australia, the establishment of a conservation covenant or purchase of a private reserve through the National Reserve System does not prevent minerals exploration or mining. This is because subsurface resources are owned by the state and are not part of a privately owned surface title. There have been recent threats to some private protected areas due to mining approvals being given by a state government, against the wishes of the private landholder (Adams and Moon 2013).

The Australian private protected area estate

Although Australia has a relatively comprehensive national database for recording the location, size and management intent (IUCN categories) of public protected areas and Indigenous protected areas, the national reporting of private protected areas is somewhat more *ad hoc* and is not comprehensive. Protected area data are compiled nationally every two years or so as part of the Collaborative Australian Protected Area Database (CAPAD) (Department of the Environment 2014). This generally involves state and territory governments providing spatial data and IUCN categories to the Australian Government which already holds data on Indigenous Protected Areas and land purchased through the National Reserve System Program, including private protected areas under this scheme. However, only some jurisdictions provide information on conservation covenants (in 2012 this was South Australia, Queensland and Tasmania). As such, gaining a comprehensive picture of the number and area of private protected areas in Australia is difficult.

I sourced data on property number and area conserved from each conservation covenanting program and major private land trusts in Australia in September 2013. If considering all 'in perpetuity' conservation covenants under a dedicated program to be private protected areas and land owned by non-government organisations and managed for the purpose of biodiversity conservation, there were approximately 5,000 terrestrial properties that could be considered private protected areas in Australia covering 8,913,000 hectares as at September 2013. This comprises almost 4,900 conservation covenants covering over 4,450,000 ha (Table 3) and approximately 140 properties owned by private land trusts covering approximately 4,594,120 ha (Table 4), and a small number of private protected areas owned by other organizations. Some of these large properties held by non-government organisations have covenants and where known these have been counted only once in deriving the total figure.

There are a number of other covenanting arrangements (or covenant-like arrangements) that may not qualify as private protected areas, but are effectively managed in the same way as other conservation covenants (Table 5). It is recognised that not all properties owned by private conservation trusts would necessarily qualify as private protected areas under the current National Reserve System criteria (mainly due to legal protection) however they are managed with this explicit intent and are moving towards greater security and many would be widely considered 'private protected areas'.

The size of private protected areas varies widely and is influenced by a number of factors, including size of historical subdivision of land parcels and amount of vegetation clearing in a region. Generally properties purchased by non-government organisations are larger than the average area covenanted by individual landowners. Covenanted land can be as small as ~1 ha while private reserves owned by non-government organisations can be in the hundreds of thousands of hectares.

In terms of total area, private protected areas make up a relatively small proportion of the overall area protected within Australia's National Reserve System, although this area and relative proportion has increased significantly since the year

Table 3. Number and area of major conservation covenanting programs in Australia (as at September 2013).

Covenanting program	Number	Total area (ha)	Average covenant size (ha)
Victoria: Trust for Nature covenants	1,242	53,370	43
NSW Voluntary Conservation Agreements	367	143,050	390
NSW Registered Property Agreements	237 †	44,150	186
NSW Nature Conservation Trust covenants	73	16,687	229
Tasmanian Private Land Conservation Program covenants	703 ‡	83,644	119
South Australian Heritage Agreements	1,518	643,631	424
Queensland Nature Refuges	453	3,438,004	7589
Western Australian (Department of Parks and Wildlife) covenants	169 §	17,386	103
Western Australian National Trust covenants	162	17,879	110
Northern Territory Conservation Covenants	2	640	320
TOTAL	4,926	4,458,441	905

Notes: † This does not include 99 Temporary Property Agreements covering ~8,450 hectares; ‡ Includes 39 covenants 'time limited' covenants covering 6,845 ha; § Number of landholders; | Area shown is area of bushland (natural habitat). Total area covenanted (included cleared land) is 64,381 ha.

Table 4. Number and area of private reserves owned by major non-profit conservation land owning organisations in Australia (as at 30 July 2013).

Organisation	Number of properties owned†	Total area (ha)	Average property size (ha)
Bush Heritage Australia	35	960,000	27,429
Australian Wildlife Conservancy	23	>3,000,000	130,400
Trust for Nature (Victoria) ‡	47	36,104	768
Nature Foundation SA	5	499,705	99,941
Nature Conservation Trust of NSW	12 §	10,182	849
Tasmanian Land Conservancy	11	7,283	662
South Endeavour Trust	7	80,646 ¶	11,506
TOTAL	137	4,518,530	

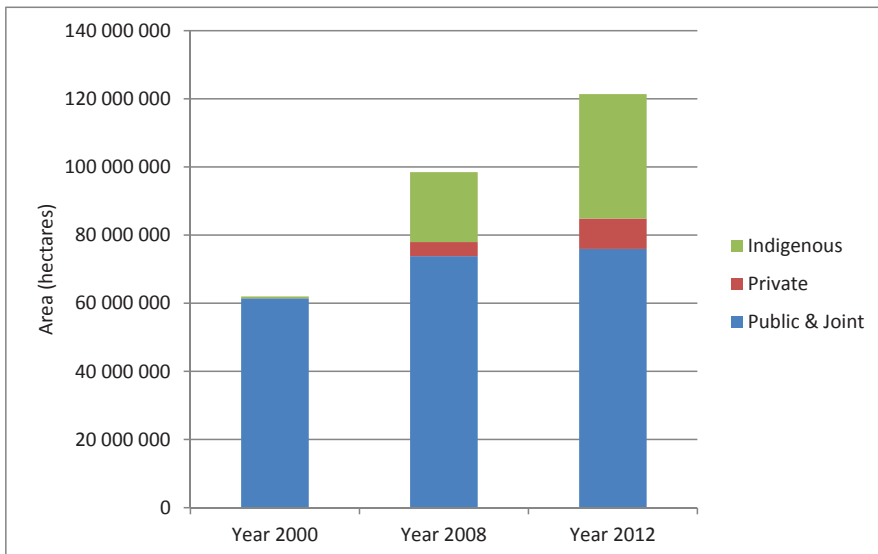
Notes: † Not all properties may have legal protection to the extent outlined earlier but all properties are effectively managed as private protected areas; ‡ In addition to this figure, 55 properties purchased by the Revolving Fund since its inception, and 52 have been on-sold, protecting 5,695 ha; § Currently holding but to be sold with covenant as part of revolving fund – a further 12 have been sold to supportive private owners, protecting 11,823 ha (included in covenant figures in Table 3; | All covenanted; ¶ The largest property, the 68,000 ha Kings Plains, is a mix of conservation and sustainable grazing.

2000 (Figures 1 and 2). As noted in above, data within CAPAD, which informs the governance types within the National Reserve System, is not complete for conservation covenants. Nonetheless, it does include most of the large private protected areas purchased with assistance from the National Reserve System program, as well as covenants from three states – South Australia, Queensland and Tasmania – which

Table 5. Conservation covenants or property agreements that due to either their level of security, allowable activities or primary intent would not qualify as private protected areas protected areas (as at September 2013).

Program	Number of agreements	Area (hectares)
Victorian covenants signed as part of BushTender under the <i>Conservation, Forests and Lands Act 1987</i>	44 †	1,500
New South Wales Wildlife Refuges ‡	672	1,890,000
New South Wales Conservation Property Vegetation Plans §	59	~6,570
New South Wales Biobanking agreements	21	3,170
Conservation covenants with the Western Australian Commissioner of Soil and Land Conservation ¶	57	5,685
'Agreement to Reserve' with the Western Australian Commissioner of Soil and Land Conservation #	441	30,880
Voluntary Conservation Agreement programs operated by south-east Queensland local governments	Unknown	Unknown

Notes: † Not all of these covenants have been completed (i.e. still in process of being put on-title); ‡ some of which are registered on the title but can be removed by the landholder; § For more information see <http://www.environment.nsw.gov.au/vegetation/pvp.htm>; | For more information see <http://www.environment.nsw.gov.au/biobanking/biobankframework.htm>; ¶ A Conservation Covenant, which is expressed to be irrevocable. The figures in the table relate to in perpetuity agreements – there are a further 46 set term agreements covering 3313 ha. Once finalised, the Commissioner does not have statutory authority to vary or discharge these covenants; # An Agreement to Reserve, which is not expressed as irrevocable. These covenants usually apply in perpetuity and may be varied or discharged by the Commissioner (there are 12 set term agreements covering 5549 ha). Thus from time to time, landowners may request the Commissioner to discharge these types of covenants. If the Commissioner refuses to discharge the covenant, there is facility under the Act to appeal the Commissioner's decision.

**Figure 1.** Increase in extent of protected areas in the National Reserve System between 2000 and 2012, including ownership type (data from the Collaborative Australian Protected Area Database 2000, 2008 and 2012 for public and Indigenous protected areas and from this paper for private protected areas).

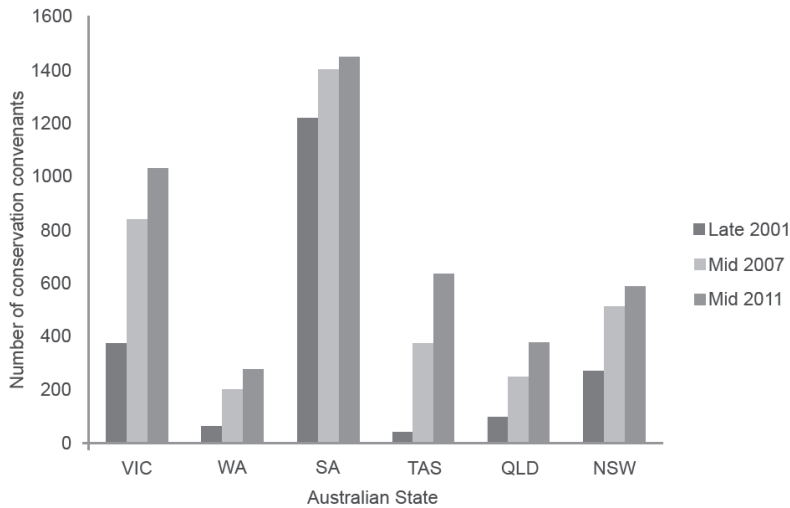


Figure 2. Number of conservation covenants in Australian States in 2001, 2007 and 2011. Note: represents covenants in programs listed in Table 3, with the exception of WA covenants in 2001 which includes those signed by AgWest (Department of Agriculture) (Stephens 2002) – these were not included in the 2007 and 2011 totals. The NSW area does not include Wildlife Refuges.

would comprise as significant majority of the total area under conservation covenant in Australia.

To address the gap in CAPAD, in 2009, the National Conservation Lands Database was compiled and included the majority of high security mechanisms operating on private land in Australia, where conservation is the sole or key objective. The data set contains all agreements from the inception of the program through which they were delivered to (and including) those established on 30 June 2009. The 2009 iteration of the database included summary statistics on number and area but, unlike CAPAD, polygon information for these covenants was not made publically accessible (see Figure 3). The objective was that this database would be updated annually but there has not been a publicly released version of the data since 2009 and it is unlikely that an update will be released in the near future.

There a number of factors that seem to be currently inhibiting this national reporting:

- 1) Privacy concerns for private landowners in revealing the location of their properties.
- 2) A lack of coordination/process between state government, Australian Government and covenanting agencies outside of the state nature conservation agency.
- 3) A lack of assessment as to whether covenants (generally or specifically) meet the protected area classification or National Reserve System inclusion criteria.

Nonetheless, each state covenanting program maintains their own database of covenants.

Ownership and occupation of private protected areas in Australia

Conservation covenants make up the majority of individual private protected areas in Australia and for most covenanted properties, people either live on or have the provision to live on the properties. In most cases it is private individuals or families that own properties with covenants over them. In many cases a covenant will be a smaller part of a larger property, such as a farm, that is not part of the protected area. In other cases there might be a specific zone within the covenant that recognises an existing or future house. Specific details about what is and what is not permitted on a covenanted private protected area is set out in the covenant document which is agreed upon by the landholder signing the covenant. Activities that might degrade the conservation value of the covenant generally are not permitted. The majority of covenants are not generally 'open access' as they are the property of a private individual and not generally dedicated for commercial purposes. For private protected areas owned by NGOs, there will often be a dedicated land manager living on the reserve, particularly in remote locations.

There are few private protected areas owned by 'for-profit groups' (companies) in Australia. A recent example is Henbury Station in central Australia, purchased by R.M. Williams Agricultural Holdings (Pearse 2012) whose intention for the property was both biodiversity conservation and carbon sequestration (by removing stock from this former pastoral station). Despite being purchased with funds from the Australian Government's National Reserve System Program, the hopes for a tradeable carbon sequestration credits from the property were not realised and the property was recently sold and less than 20 per cent will be formally protected within a conservation covenant (Brann and Brain 2014). Earth Sanctuaries Ltd was the first publicly listed company in Australia to have wildlife conservation as its primary goal, owning 11 private reserves covering c. 100,000 ha at its peak of land ownership (these properties would not have technically qualified as private protected areas under the current terminology, but were effectively managed with this intent). Earth Sanctuaries sought to generate income by placing a monetary value on the threatened species it owned (Sydee and Beder 2006). Yet, the company overestimated the revenue-generating potential of its extensive landholdings and suffered financial difficulties and was eventually delisted in 2006. The majority of its reserves were purchased by the Australian Wildlife Conservancy, but the demise created a potential loss in confidence in the private nature reserve system in Australia (Fitzsimons and Wescott 2002).

Ownership of private protected areas can change in a more deliberate way. For example, a number of private land trusts operate revolving funds whereby a property is purchased by the NGO and then on-sold with a conservation covenant attached. For example the Queensland Trust for Nature has protected more than 100,000 hectares of land in Queensland having acquired eleven 14 properties and sold 8 to private land owners with Nature Refuge agreements attached to title (Queensland Trust for Nature 2013). Private land trusts can also transfer private reserves into the public protected area estate: for example the Trust for Nature (Victoria) has transferred 65 properties to the Victorian Government in total comprising 6,745 ha.

There have been a smaller number of acquisitions by community groups, such as the Twin Creeks Community Conservation Reserve (Department of the Environment 2013). There are also emerging hybrid models of private protected areas with other governance types. For example Fish River Station in the Northern Territory was purchased by the Indigenous Land Corporation with financial support from the Australian Government's National Reserve System program and two NGOs, The Nature Conservancy and Pew Environment Group (Fitzsimons and Looker 2012). It is a private protected area, but will be handed back to the Traditional Owners in the future. On Cape York, a consolidated program of land acquisition and tenure resolution of public land has seen the delivery of 580,000 ha of new national parks, and 703,000 ha of Aboriginal land, of which 90,000 ha are managed as Queensland Nature Refuges (conservation covenants) (Leverington 2012).

Almost all marine waters in Australia are owned by the Crown (government) and there are no private protected areas in the marine environment.

Main management approaches and IUCN categories

For public protected areas in Australia, IUCN categories are determined by the jurisdiction which manages the protected areas, primarily the state/territory governments. This is often done in accordance with guidance from state level documents (e.g. Department of Natural Resources and Environment 1996), the *Draft Australian Handbook for the Application of IUCN Protected Area Management Categories* (WCPA Australian and New Zealand Region 2000) and more recently the revised international guidelines (Dudley 2008). These data are compiled nationally every two years or so as part of the Collaborative Australian Protected Area Database. The application of these categories to private protected areas has been a somewhat more *ad hoc* approach. An analysis of CAPAD 2010 reveals that South Australia classified all their Heritage Agreements (conservation covenants) as category III (although Leaman and Nicolson (2012) suggested they are reported to the Australian Government as category VI), Queensland as category VI (with the exception of a small number as category II) and Tasmania a mix of categories Ia and VI.

For conservation covenants, the National Conservation Lands Database noted that many agencies were not confident that their interpretation of an IUCN category for their agreements was consistent with a national approach and some agencies assessed each covenant individually while others coded all agreements of a particular type the same way.

For purchases made under the National Reserve System Program, early advice from the Australian Government's environment department to non-government organisations purchasing private conservation lands was to assign private reserves as category IV. However, a review of private conservation lands in Victoria suggested that private protected areas could potentially fall in any of the IUCN protected area management categories (Fitzsimons 2006). Indeed a recent purchase of the

180,000 ha Fish River Station in the Northern Territory has seen this property classified as category II (Fitzsimons and Looker 2012) and other land acquisitions in Gondwana Link corridor are also classified as IUCN category II (Bush Heritage Australia 2013).

The current application of IUCN protected area management categories to private protected areas in Australia is in need of review, as is a national discussion of the implications of the classifications. Although the National Reserve System Strategy (Natural Resource Management Ministerial Council 2009, p. 4) recognised the need for “consistent approaches informed by the development of national frameworks for management effectiveness and protected areas on private lands”, little progress has been made to date. The formation of the Australian Land Conservation Alliance (<http://www.alca.org.au/>), made up of the main covenanting land trusts and The Nature Conservancy will seek to engage discussion on topics such as this.

The distribution and landscape context of private protected areas in Australia

Up until the mid-1990s, the public protected area system in Australia was typically created from existing public land, which itself was often the ‘left overs’ from land not suitable to use for agriculture. Typically this was steep and forested country or marginal desert country (Pressey and Tulley 1994; Pressey et al. 1996). The advent of the National Reserve System Program and scientific principles of comprehensiveness, adequacy and representativeness saw a much more targeted approach to reserve creation, with an emphasis on filling gaps and targeting the inclusion of under-represented ecosystems (Fitzsimons and Wescott 2004). The role of conservation non-government organisations is considered by the Australian Government as “critical, as they complement the public reserves by filling conservation gaps, purchasing or covenanting land where governments are unable to do so” (DSEWPC 2013). The Natural Resource Management Ministerial Council also recognise that many threatened species and under-represented communities occur on private land that is not for sale, but that farmers and graziers are increasingly placing voluntary, in perpetuity covenants on their property.

Most conservation covenanting programs were established before the concepts of comprehensiveness, adequacy and representativeness were explicit in conservation policy in Australia. Nonetheless, in a review of conservation covenanting programs in 2007, Fitzsimons and Carr (2014) found that most programs now seek to complement the comprehensiveness, adequacy and representativeness of the public reserve system, either stating so explicitly or by aiming to protect the highest priority ecosystems on private land.

Gilligan and Syneca Pty Ltd (2007) found that the Tasmanian Private Forest Reserve Program, one of the few covenanting programs where financial payments were made to landholders to secure new covenants, “made a significant contribution to achieving the conservation outcomes set out in the Tasmanian Regional Forest Agreement by

securing in perpetuity more than 40,000 hectares of private forests targeted in the Strategic Plan for the Program” (see also Iftekhar et al. 2014).

However, it should be recognised that covenants are generally established for a range of reasons beyond just complementing the comprehensiveness, adequacy and representativeness of the reserve system. It is often the landholders themselves that approach a covenanting agency to have a covenant placed on their property to ensure the natural assets on their property are protected when the property is sold or passed down to their heirs. Fitzsimons and Wescott (2001) found that there were clusters of small covenants (and other less secure private land conservation mechanisms) on the vegetated outskirts of larger regional population centres in Victoria. More recently, the Trust for Nature (2013) has shown how a more targeted approach to covenant establishment has significantly increased the proportion of covenants in under-represented bioregions.

New private protected areas may also be established with the explicit aim of buffering (Coveney 1993) or linking (e.g. Bradby 2013) existing protected areas. Fitzsimons and Wescott (2005) and case studies within Fitzsimons et al. (2013a) highlight the catalysing role of land purchase by non-government organisations in establishing new connectivity conservation initiatives in a region.

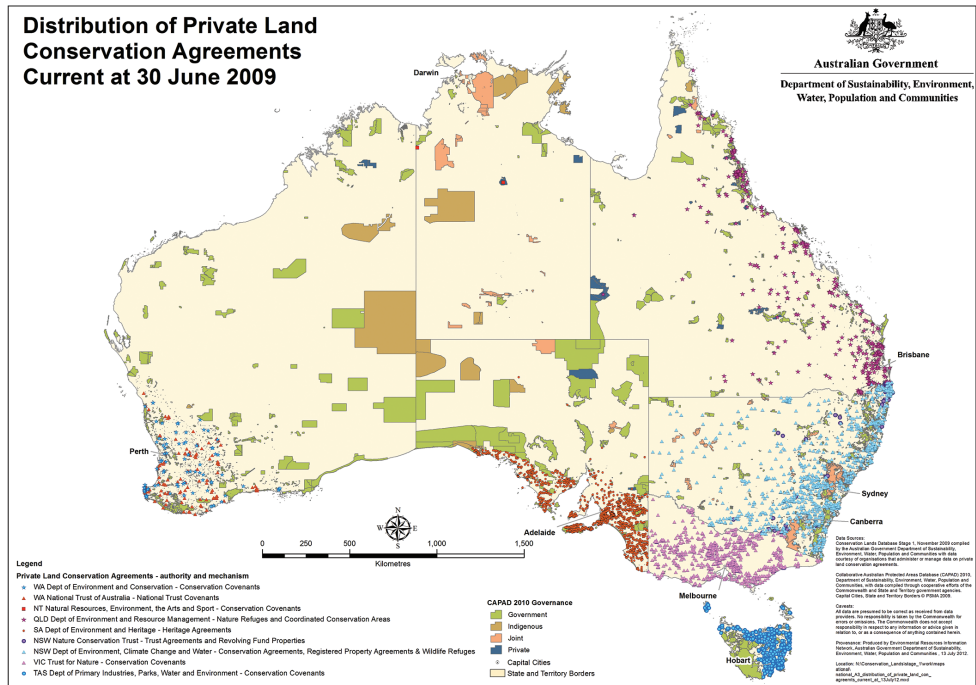
In a number of state jurisdictions, covenanting of leasehold land, which makes up a significant proportion of inland Australia, is significantly harder than covenanting freehold land (due to conflicts in management intent and required use of land between covenant and pastoral lease legislation). This means that at a national level covenants are more skewed towards freehold properties in eastern and southern Australia and Tasmania (Figure 3).

Incentives for establishment and maintenance of private protected areas

There is a range of incentives offered for private land conservation, including the establishment of private protected areas, however these differ across the country and differ within states. For non-government organisations purchasing land a significant financial incentive to establish new private protected areas was provided by the Australian Government through the National Reserve System Program, which offer two-thirds of the purchase price (the National Reserve System Program had a dedicated fund for land acquisition from the mid-1990s up until December 2012 when it was not renewed – Fitzsimons et al. 2013b).

At a national level, tax concessions are available to land owners who enter into conservation covenants (with an approved covenanting program) to protect areas of high conservation value. To qualify for an income tax deduction all of the following conditions must be met (DSEWPC 2012):

- The covenant must be entered into on or after 1 July 2002.
- The covenant must be entered into over land which the landholder owned – leased property is not eligible.



this mechanism along with the changes in income tax marginal rates has resulted in lower incentives for a group of donors who own land, but who may have a low income. Land-rich, cash-poor landholders will not realise the full value of the tax deductibility as will a more affluent landholder. Anecdotal evidence suggests the low uptake of land-owners seeking a tax concession for any of loss in value on their property as a result of the covenant was in part due to the costly and bureaucratic nature of the valuation with little guarantee of a real loss in property value. This provision is also inconsistent with the broad message given by covenanting programs that a covenant does not usually result in a loss in property value (see Fitzsimons and Carr 2007)."

Property rates are charged by local governments in Australia and some local governments offer a partial or full rate rebates for covenanted properties. This rate relief varies significantly across the country and within states. There has been a significant increase in incentive payments, to encourage the signing of covenants in high priority, under-represented bioregions in the past decade (Adams et al. 2014). Where there are open calls or tenders for funding conservation activities on private land within a region, covenants will often receive a higher priority over shorter-term conservation agreements, all else being equal. However, within the last decade there has been a focus on stewardship payments for shorter-term (e.g. 5 to 15 years) management agreements (Wardrop and Zammit 2012). Further research is needed to determine if certain landholders are less likely to sign up to long-term covenants even if incentive payments are available.

Reporting and measures of conservation or management effectiveness

Requirements of owners of private protected areas to report on their activities vary. As a condition of funding for land acquisition (such as through the National Reserve System Program) or management (such as through various stewardship payment programs), reporting is required.

For private protected areas purchased with funding from the National Reserve System program, the 'Funding Deed' requires Monitoring, Evaluation, Reporting and Improvement (MERI) plans be prepared for each property (Australian Government 2013). In addition to twice-yearly progress reporting, a final report is required at the completion of all tasks associated with setting up the land as a protected area and preparing for its long-term management. As National Reserve System Program land purchase projects have similar reporting requirements and a reasonably standard set of activities, a number of templates have been prepared. These templates and reports have a number of purposes, including:

- to report on key milestones and activities throughout the course of the project and to provide updated documentation relating to formalising the land as a protected area;
- to describe the contribution of the project to the comprehensiveness, adequacy and representativeness principles of the National Reserve System;

- to evaluate the effectiveness of the methodology and approaches used to establish the project as a protected area and to prepare for its long-term management; and
- to incorporate lessons learned into future work in the project and in the National Reserve System land purchase program.

If conservation covenants have received funds as part of covenant establishment, owners will typically have to report on the annual activities and outcomes. For those established without financial assistance the level of reporting required and stewardship capacity from the covenanting agency varies. In Victoria, as part of the Trust for Nature's Stewardship Program monitoring of conservation covenants is undertaken at least once every five years and reported in a stewardship report (Trust for Nature 2014). Management plans are written by Trust for Nature regional managers and or stewardship officers, in consultation with the landowners.

In a review of conservation outcomes of conservation covenanting programs across Australia, Fitzsimons and Carr (2014) found that the role of monitoring and types of monitoring varied widely. For example, monitoring programs ranged from the basic statewide to regional inventories, such as number and area of covenants and increase in growth in signing covenants per year, through to assessments of the contribution that covenants are making to the conservation estate at the bioregional level (e.g. enhancing representation and/or improving linkages in the landscape or buffering protected areas). Other monitoring measures included site-based assessments such as complying with the conditions of the covenant and various forms of ecological monitoring. Some programs did all of these, whereas others only undertook the broader assessment. In terms of on-ground ecological monitoring, the techniques and emphasis between programs varied and the purpose for doing this was more to inform management than to necessarily gain quantifiable ecological data suitable for statistical analysis. Some were using methods that were consistent or comparable with what was being used in the rest of the jurisdiction (i.e. elsewhere with the state nature conservation agency/parks service), unlike others that had a more simplified or more advanced version of what is used elsewhere in the state.

Some covenant programs had collected benchmark ecological information for most covenants at the time of signing and most programs now undertake this on the signing of new covenants. Site visits ranged from yearly to five-yearly or on an 'as-needs' basis. A lack of resources to monitor (staff numbers and time), knowing what to monitor, inconsistent monitoring methodologies, lack of benchmark data and length of time to see meaningful results from monitoring, were all considered potential barriers to evaluating the biodiversity conservation outcomes of conservation covenants (Fitzsimons and Carr 2014).

Future directions and challenges for private protected areas in Australia

As outlined above and elsewhere (e.g. Gilligan 2006), private protected areas are making an increasing contribution to the area and ecosystems conserved in Australia.

However, the sector faces some unique challenges which will need to be addressed if private protected areas are to achieve their full potential. Some of the most significant challenges and opportunities are outlined below:

Managing ecosystem processes: Like managers of public and indigenous protected areas, managers of private protected areas face challenges in managing ecosystem processes on their property that are often outside of their direct control (e.g. environmental flows for wetlands or floodplain ecosystems) or may be difficult due to the size of the property or capacity of the landowner (e.g. application of ecological burns; Halliday et al. 2012). Recognising this, a number of the non-government organisations have established programs that go beyond their property boundaries to manage processes and threats such as fire (Legge et al. 2011), pest plants and animals (Walsh et al. 2013), and improve connectivity (Edwards and Fox 2013) in the surrounding landscape. However, individual covenantors will have limited capacity to do this, and cooperation and alliances with government agencies, surrounding landholders and other groups not normally associated with conservation will be crucial.

Tenure reform and increased security for protection mechanisms: Most of the large private protected areas purchased for conservation by non-government organisations in north or central Australia occur on pastoral leases. This means that a) the primary purpose of the lease is not likely to be for conservation, b) placing a protective conservation covenant on the lease may be problematic due to an inherent conflict between the purpose of the lease and that of the covenant and c) some cattle or sheep grazing may be legally required regardless of whether this is ecologically desirable. Although some state governments do not enforce the pastoral conditions (or may insist on only a minimal area to be grazed), considering the Australian taxpayers through the National Reserve System program have paid two-thirds of the purchase price for the majority of these large properties, improved protection arrangements, tenure reform or both are required to ensure the security of these conservation investments into the future.

Reinstating a National Reserve System program with a dedicated fund: For the first time in almost two decades the Australian Government's National Reserve System Program, comprising a dedicated funding allocation and specialist policy and administrative unit was discontinued in late 2012. This program and associated policies were fundamental for driving significant strategic growth in Australia's protected area estate, on public, private and Indigenous land tenures. Taylor et al. (2014) believe it is highly unlikely that Australia can achieve its long-standing commitments to an ecologically representative National Reserve System without a reinstatement of this funding. Loss of a dedicated funding program will slow the growth of the private land trust sector for two reasons. Firstly, there is a need to be able to access funds quickly when desirable land comes on to the market. Secondly, the leverage model the National Reserve System encouraged was particularly popular with philanthropists as they saw their gift being matched by government. Other funding mechanisms such as smarter use of the substantial investments in offsets for development will also need to be considered if the private land trust sector is to continue to grow.

More consistent incentives for covenantors: As highlighted above, there is substantial variation in the types and amounts of financial assistance offered to covenantors

between, and even within, Australian jurisdictions. Some of this variation is justified, such as governments providing targeted payments for the establishment and management of under-represented ecosystems to meet national and international targets, often through tender-based approaches. However, in order to recognize the role covenants are playing financially in protecting biodiversity and to legitimize this land use further, ensuring greater consistency in the rate relief offered to covenants and providing tax deductibility for conservation management activities (similar as for those provided to primary producers) should be a priority for all levels of government.

Access to new markets for funding. Until recently, biodiversity and ecosystem services have largely been taken for granted. However, their value is increasingly recognised and payments for ecosystem services are emerging in Australia (Figgis et al. 2015). Some owners of private protected areas have already taken advantage of this. For example, the owners of Fish River Station are paid to implement traditional fires and reduce carbon emissions (Walton and Fitzsimons in press). However, there remains a distinct possibility that the majority of existing private protected areas will not be able to enter into some new payment for ecosystem service markets. This is because the 'additionality' they offer will be difficult to prove when they are already considered to have legally protected the ecological assets on their properties. Careful consideration of policy will be required to ensure those choosing to have their properties protected are not excluded from these markets and left potentially financially worse off than those participating in the markets, but choosing not to protect their properties. If not addressed this could create a significant disincentive for landholders considering entering into conservation covenants into the future.

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Food supplementation does not increase demographic rates in a passerine species of conservation concern

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Abstract

Numerous studies have examined the effects of the provision of supplementary food on aspects of avian reproductive success, but far fewer have gone on to examine the potential positive effects of food supplementation on the demographic rates which are key for population growth rate. Testing for potential effects of food shortage on vital rates is likely to be particularly important in species of high conservation concern, where populations are particularly small, isolated or decreasing rapidly. Here we test the effects of the provision of supplementary food on reproductive success, body condition at fledging and post-fledging survival of ring ouzels (*Turdus torquatus*), a species of high conservation concern in the UK. However, food supplementation had no detectable effect on any of these parameters. There was no significant difference in return rates of fed and unfed fledglings in the year following hatching, and most post-fledging mortality was apparently caused by predation by raptors and mustelids. We conclude that the supply of invertebrate food sources for nestlings was not a major limiting factor in our study area, at least during this two-year study. Further studies are required to quantify the precise mix of habitats used by ring ouzels, at the appropriate scale, which provide concealment from predators and access to food supplies throughout the spring and summer months.

Keywords

Food supplementation, demographic rates, passerine species, conservation

Introduction

The provision of supplementary food has been trialled as a method for increasing reproductive success and/or survival in a range of avian species, especially those of conservation concern where populations are particularly small, isolated or decreasing rapidly (e.g. Armstrong et al. 2007; Houston et al. 2007; Schoech et al. 2008). However, such trials have had mixed success, with some showing apparent positive impacts of food supplementation, and others showing mixed or uncertain results. For example, supplementary feeding increased kakapo (*Strigops habroptilus*) and hihi (*Notiomystis cincta*) reproductive success, through improved nestling survival to fledging (Elliott et al. 2001; Walker et al. 2013). However, food supplementation did not result in increased survival of adult bearded vultures (*Gypaetus barbatus*), despite increasing the survival of pre-adults, and did not influence adult or fledgling survival in barn owls (*Tyto alba*) (Meek et al. 2003; Oro et al. 2008). Thus, further data on the effects of food supplementation on avian reproductive success and survival, especially in species of conservation concern, is desirable.

Since providing supplementary food can be a costly and labour-intensive conservation management action, it is important that there is clear evidence for food shortage before feeding begins. For example, in the New Zealand hihi, providing supplementary food increased abundance and survival of translocated birds on Kapiti Island, following evidence that food was limiting on previous island translocations (Chauvenet et al. 2012). It is also important that the life-history stage(s) which are most likely to influence population growth rate (λ) of the study population are targeted for conservation action. For instance, numerous studies have shown that avian λ is more heavily influenced by first year and adult survival than by reproductive success (e.g. Sæther and Bakke 2000; Reid et al. 2004; Schaub et al. 2006). Thus, there would be little purpose in providing supplementary food to increase reproductive success, if this demographic rate was predicted to have little impact upon λ in the target species.

The ring ouzel, hereafter ‘ouzel’, is a species of high conservation concern in the UK (Eaton et al. 2011). Declines of this migratory, typically double-brooded, upland thrush species are widespread throughout the UK, but the underlying mechanisms are poorly understood (Sim et al. 2010). However, in a study population in Scotland, a decline in brood size at fledging of successful early season nests during 1998–2009 suggested that poor food supply to nestlings may have reduced survival from hatching to fledging (Sim et al. 2011). In addition, post-fledging survival was higher for individuals fledging from broods earlier in the season, although there was also support for models which included a positive effect of body condition on post-fledging survival (Sim et al. 2013b). Thus, reduced body condition at fledging may have been caused by food shortage, resulting in lower survival during the post-fledging period. Ouzel nestlings are fed mainly on earthworms (*Lumbricidae*), larval leatherjackets (*Tipulidae*) and ground beetles (*Coleoptera*) (Burfield 2002), and it is likely that the abundance of these invertebrate groups has declined in the UK uplands in recent decades due to changes in moorland management, such as intensification of heather burning and increased ditch construction, which might affect soil conditions, vegetation structure and com-

position (Dennis et al. 1998; Buchanan et al. 2006; Evans et al. 2006). Therefore, ouzel declines may be linked to declines in the abundance of key invertebrate prey.

A recent demographic analysis of one ouzel study population indicated that λ was most sensitive to apparent first-year survival (fledging to age one year), closely followed by re-nesting rate and early-season reproductive success, and that first-year survival contributed most to observed variation in λ (Sim et al. 2011). Additionally, most (63–68%) estimated first-year mortality occurred in the first five weeks post-fledging, indicating that low juvenile survival through this specific period may substantially affect λ (Sim et al. 2011). Therefore, conservationists would be most likely to improve ouzel λ by implementing measures that increase reproductive success and post-fledging survival.

In this paper we report the results of a two-year field experiment in which we provided supplementary food to a declining population of ouzels, in an attempt to improve their reproductive success and post-fledging survival. We hypothesised that adult provisioning rates to nestlings, fledging success, brood size at fledging, nestling body condition at fledging and post-fledging survival would all be higher, and within-brood variation in nestling body condition would be lower, in fed than in control territories, where no supplementary food was provided.

Methods

Study area and species

The ring ouzel is a medium-sized migratory thrush that breeds in north-west and central Europe and winters in southern Spain and North Africa (Wernham et al. 2002). We studied a population in Glen Clunie (56°56'N, 3°25'W; 18 km²), north-east Scotland during 2011–12, that has been part of a long-term study since 1998 (Sim et al. 2011). An additional area of approximately 8 km² in the adjacent Glen Callater (56°57'N, 3°22'W) was included in the study in 2011 only. The vegetation in the upper slopes of these glens is a mosaic of heathers (*Ericaceae*), bilberry (*Vaccinium myrtillus*), crowberry (*Empetrum nigrum*) and rough grassland, interspersed with numerous crags, scree slopes and gullies, and improved grassland and small conifer plantations in the valley floors. The area is managed for sport shooting of red grouse (*Lagopus lagopus scoticus*) and red deer (*Cervus elaphus*), largely through heather burning, used to provide a mosaic of vegetation structure thought to benefit red grouse. The area is also grazed by sheep and cattle at low densities. Gamekeepers legally and effectively eliminated generalist predators such as red fox (*Vulpes vulpes*) and carrion crow (*Corvus corone*), but we regularly observed stoats (*Mustela ermine*) and weasels (*M. nivalis*).

In each year we aimed to locate all ouzel breeding pairs and nesting attempts. The study area was systematically surveyed, by walking all ground to within 200 m of observers, every one to two weeks between mid-April and mid-July (Sim et al. 2011). Ouzels vigorously defend the immediate vicinity of nests, but disputes outside this area are rare (Burfield 2002). A successful breeding cycle requires 29–30 days (16–17 days for egg laying/incubation plus 13 days until fledging; Burfield 2002), and Brit-

ish ouzels regularly make two, rarely three, breeding attempts per season (Sim et al. 2012). The number of ‘early’ and ‘late’ pairs were defined as those laying eggs within 30 days of the first laying date recorded in the population, and on or after day 31, respectively, in each year (Sim et al. 2011). Egg laying date was recorded directly in nests found during laying or hatching, estimated using known relationships between chick age and wing-length and mass (Burfield 2002) or by comparison with photographs of known-age nestlings. We visited nests every 3–5 days to record potential breeding failures, to count the number of eggs and hatched chicks, and to measure the mass and wing length of chicks. During 2011–12, we located 69 early and 49 late nests before nestlings fledged, comprising 93% and 98% of known successful early and late nests, respectively. We located 77% of nests with eggs and 23% with nestlings.

To allow individual identification, 263 nestlings (145 from early and 118 from late nests), comprising 90% of individuals known to fledge in the study area, were ringed with BTO metal rings and individual combinations of three plastic colour rings. In addition, 17 (2011) and 29 (2012) individually colour ringed adults marked previously in the study area as either breeding adults or nestlings returned to breed, and a further 12 (2011) and 24 (2012) adults were caught and colour ringed.

Experimental design

In 2011, we randomly allocated 50% of the known territories (i.e. those occupied at least once between 1998 and 2010; $N_{\text{total}} = 86$) to receive supplementary food for both early and late breeding attempts (‘fed’ territories). The remaining 50% of the known territories were ‘control’ territories, where no supplementary feeding occurred. In 2012, we reversed treatments, so that control territories from 2011 became fed territories, and vice versa. In both years our aim was to have approximately equal numbers of fed and control territories. However, this ‘ideal’ experimental design was not possible in territories that were occupied only in either 2011 or 2012, in those territories where ouzels did not find or utilise the supplementary food, or where feeding had to be abandoned before nestlings fledged due to other species taking the supplementary food (see Results for details). When feeding was not possible in a planned fed territory, the next occupied territory on the ‘fed’ random list was selected to receive supplementary food. Thus, despite our ambitions, we ended up with an unbalanced experimental design in terms of numbers of fed and control territories across the two years.

Food supplementation

Supplementary food was provided during the ouzel nestling-rearing stage in black plastic seed trays (38 cm × 24 cm × 6 cm) placed on prominent knolls, boulders or in short grass-rich areas on the ground, between 20 m and 50 m from ouzel nests. These locations were chosen to make the food as obvious as possible to the ouzels, while reducing the risk of

predators locating the nest. We observed these feeding trays to determine (a) if the ouzels and/or other species fed from them and (b) what food source [live earthworms (*Dendrobaena* spp.), or live mealworms i.e. the larvae of the mealworm beetle (*Tenebrio molitor*)] was preferred. In 2011, we provided 100 g of both earthworms and mealworms in each territory daily, in order to replicate the key natural food sources of nestling ouzels. Mealworms have a relatively high protein (45–60%) and fat (30–45%) content (Bernard et al. 1997; Tran et al. 2013), similar to that found in beetle larvae (protein 23–66%; fat 9–67%; Banjo et al. 2006; Van Huis et al. 2013), and we therefore assumed that they provided a suitable partial food source for nestling ouzels. However, since only 5 of 13 (38%) ouzel pairs selected earthworms in 2011 and these escaped relatively easily from the feeding trays, thus becoming unavailable to ouzels, only 100 g of mealworms was provided daily in 2012. Mealworms were regularly taken by ouzels, and since they were unable to escape from the trays they were available for longer periods. At a single territory, we observed a male ouzel from a control territory taking supplementary food from the neighbouring fed territory. Feeding trays were only very rarely completely empty when they were re-visited to replenish them with mealworms (i.e. we effectively provided supplementary food *ad libitum*).

Provisioning rates

We monitored adult provisioning rates to nestlings at fed and control sites using $\times 10$ binoculars or $\times 15$ – 40 zoom telescopes, from hides, cars, or by observers well concealed in open moorland, at distances of 30–200 m from nests, depending on topography and the sensitivity of the adults to disturbance. Each provisioning watch lasted for 60 minutes, and we classified food brought to the nest as supplementary (earthworms or mealworms taken from the feeding trays) or natural (gathered away from the feeding trays and therefore assumed to be natural). In 2011, we carried out 1–2 (mean 1.25 ± 0.11) watches at control nests when nestlings were aged 7–12 (mean 9.37 ± 0.33) days old, and 1–8 (mean 2.60 ± 0.22) watches at fed nests when nestlings were aged 5–13 (mean 9.26 ± 0.25) days old. In 2012, we carried out 1–3 watches at control (mean 1.97 ± 0.10) and fed (mean 1.95 ± 0.10) nests when nestlings were aged 4–12 (mean; control 8.16 ± 0.30 ; fed 7.94 ± 0.31) days old.

Body condition at fledging

To test whether within-brood variation, and individual nestling body condition, at fledging varied between fed and control nests, we measured body condition index (BCI) as the residual of a regression of body mass on wing length³ (body mass = $66.2 + 0.00001 \times \text{wing length}^3$; $r^2_{\text{adj}} = 0.05$; Genevois and Bretagnolle 1994). We measured wing length (maximum chord to the nearest 1 mm) and body mass (to the nearest 0.5 g, using a Pesola balance) for all nestlings aged 10–14 (mean for control and treatment nestlings = 12) days post-hatch (Burfield 2002).

Juvenile post-fledging survival and return rates

To measure survival during the post-fledging period, we fitted nestlings with 1.8 g TW4 single-celled radio transmitters (Biotrack 2013) just prior to fledging (10–14 days post-hatch), at which age we also recorded brood size (the number of nestlings alive). Transmitters were thigh-mounted using a silastic (pliable plastic) harness with a weak cotton link, to allow tag loss prior to migration (Hill et al. 1999; Rappole and Tipton 1991). In 2011, we fitted transmitters to 11 fed and 25 control nestlings from early broods, and 14 fed and 16 control nestlings from late broods. In 2012, we fitted transmitters to 17 fed and 23 control nestlings from early broods, and 19 fed and 18 control nestlings from late broods. Individuals fitted with transmitters appeared to fly as well as juveniles without transmitters when flushed, and dispersed over similar distances and time periods as colour ringed siblings. They thus showed no detrimental behavioural effects of radio transmitters, in common with previous studies (Rae et al. 2009; Gow et al. 2011; Sim et al. 2013b). We aimed to fit transmitters to at least one nestling from all successful broods. Each nestling in the two broods with a single nestling were fitted with transmitters. In the remaining 77 broods of 2–5, nestlings were ranked according to maximum chord wing length (the best predictor of nestling age; Burfield 2002), and 1–3 nestlings were selected at random according to these rankings, and fitted with transmitters. Thus, transmitters were fitted to single nestlings in 17 broods of 1–4, two nestlings in 57 broods of 2–5 and three nestlings in four broods of 5.

We tracked juveniles with transmitters and recorded their approximate locations every 3–4 days post-fledging, until the individual was found dead, shed the transmitter, or disappeared and was assumed to have dispersed from the study area. Individuals were tracked at different times on different days. Transmitters had signal ranges of approximately 10 km when in direct line of sight, but more typically 2–3 km depending on terrain, and a battery life of 3–4 months. We used Advanced Telemetry Systems (ATS) scanning receivers attached to car roof-mounted aerials to provide approximate locations. Hand-held Telonics TR-4 receivers, attached to three-element Yagi antennas, were used to visually locate each individual on foot, and record their location using a Garmin Global Positioning System (GPS) 12 Personal Navigator. In addition, we recorded the observed return rates of individually colour-ringed nestlings from fed and control nests in the years following ringing.

Causes and timing of mortality

We examined remains of dead juveniles to determine the most likely cause of death. Individuals found in raptor nests or elsewhere with plucked feathers and bent radio-tag aerials were assumed to have been eaten and most likely killed by raptors, whereas those located underground in tunnels, under boulders, or in the open with bitten feathers and straight aerials were assumed to have been eaten and most likely killed by mammals (Thirgood et al. 1998). We assumed that juveniles with no apparent injuries

found close to their nest soon after fledging had died from starvation or exposure. Because juvenile location and status (dead/alive) were checked at 3–4 day intervals, the timing of mortality was also accurate to within 3–4 days.

Statistical analysis

We were primarily interested in the level of support for supplementary feeding on ouzel fledging success (the proportion of hatched nestlings that fledged, excluding nests which failed to fledge any young since these were almost certainly predated), brood size at fledging (again excluding nests which failed to fledge any young), nestling BCI at fledging (both for individual nestlings and within-brood variation in nestling BCI, the latter measured as the standard deviation of brood BCI) and post-fledging survival, compared to control territories (0 days feeding). However, since there was considerable variation in the number of days that nestlings received supplementary food (hereafter ‘feeding days’; see Results for details), we used feeding days as a predictor variable in all analyses, rather than the binary predictor fed/control.

We used Generalised Linear Models (GLMs; in the base package in R; R core team 2013), and Generalised Linear Mixed Models (GLMMs; in the LME4 package in R (Bates et al. 2014) to assess the level of support for models predicting adult provisioning rate to nestlings, fledging success and brood size and BCI at fledging. Akaike’s Information Criterion, adjusted for small sample size (AICc), was used to identify the best supported model that included the parameters of interest (Burnham and Anderson 2002). The model with the lowest AICc is the best supported model, and provides the best fit to the data of the models compared (Burnham and Anderson 2002). We examined models with ΔAICc values of ≤ 2.0 to see if they differed from the best supported model by one parameter, and had essentially the same values of maximised log-likelihood as this model. In such cases, the larger model is not really supported since it is ‘close’ only because it adds one parameter to the top ranked model and therefore will be within 2 ΔAICc units of it (Burnham and Anderson 2002; Arnold 2010). Similar caveats were applied to models with two extra parameters that fell within approximately 4 ΔAICc units of the best supported model. In general, we included only those variables which previous studies have found to influence ouzel reproductive success, such as brood (early or late), hatch date and fledge brood size (Burfield 2001; Sim et al. 2011). Year was included as a nuisance factor in all models. Following recommendations by Burnham and Anderson (2002), we constructed biologically plausible models with as few explanatory variables as possible to avoid over-parameterization. This meant that we did not run all possible permutations of the global model. In addition, due to relatively low sample size, we included only those 2-way interactions for factors considered to influence ouzel breeding success from the literature. Finally, we did not implement model averaging when two or more competing models had similar levels of support ($\text{AICc} < 2.0$), as the presence of any 2-way interactions makes the interpretation of model averaging problematic (Grueber et al. 2011). We first measured the rela-

tive support for models predicting provisioning rate, fledging success, and brood size and BCI at fledging that included univariate effects of brood (early or late), hatch date and hatch date², which were highly correlated (Spearman rank correlation between brood and hatch date $r = 0.86$, $p < 0.0001$). In all models, brood received the strongest support, so hatch date and hatch date² were omitted from further analyses. For the provisioning rate model, we tested for effects of factors year and brood, and covariates feeding days, brood size and nestling age on their own, when added to one another and including all possible two-way interactions. This process was repeated for the fledging success (excluding nestling age and brood size), brood size at fledging (excluding nestling age and brood size, but including brood size at hatching) and BCI at fledging (excluding nestling age) models. We modelled fledging success and brood size at fledging using a binomial error structure with a logit link function, and a Poisson error structure with a log link function, respectively. For adult provisioning rate to nestlings, individual nestling BCI at fledging and within-brood variation of nestling BCI, we used a normal error structure with an identity link function. For the adult provisioning rate to nestlings, and individual nestling BCI at fledging models, we accounted for multiple measures of nestling BCI, and multiple provisioning watches, from within the same territory by specifying territory ID as a random factor in the GLMM.

We ran juvenile survival analyses over 100 days post-fledging (25×4 -day periods), after which no individuals fitted with radiotransmitters remained within the study area, using the known-fate model in program MARK 5.1 (White and Burnham 1999). A goodness-of-fit test is not necessary for known-fate models, since the saturated model by definition fits the data perfectly (Cooch and White 2008). Known-fate models assume (a) that the fate of each individual is known with certainty (although ‘missing’ individuals can be censored during the period they are not found, and re-enter the dataset when they are relocated) and (b) that the fates of individuals are independent. When individual survival probabilities are not independent, estimated effects are unbiased, but variances are biased downward because of extra-binomial variation or overdispersion of the data (Tsia et al. 1999). Biologically, overdispersion might be expected if the fates of different brood members covary. We tested for significant violation of the assumption of independence by estimating overdispersion (\hat{c}) as χ^2/df , where χ^2 is the summation of partial chi-square values ($[\text{observed} - \text{expected}]^2/[\text{expected}]$) calculated for each possible outcome of losses within tagged brood sizes of one, two, or three young (nine possible outcomes). Expected values were calculated as: $(n/r)\varphi^r(1-\varphi)^{n-r}$, where n is brood size, r is the number of young surviving to independence, and φ is the survival rate between fledging and independence (Wiens et al. 2006). There was no evidence of overdispersion ($\chi^2 = 7.26$, $\text{df} = 8$, $p = 0.51$), and individual fates were therefore treated as independent. Twenty-seven individuals were temporarily censored for up to 13 consecutive four-day periods for which they were unobserved before being relocated.

Using AICc, we first tested the relative support for models where survival was constant or varied across all 25 four-day periods. We then tested for effects on survival of the factors year and brood, and covariates BCI, brood size and feeding days on their own, when added to one another and including all possible two-way interactions.

Results

Food supplementation

The proportion of ouzel pairs in fed territories that we observed feeding the supplementary food to their nestlings did not differ between 2011 (13/19, 68%) and 2012 (16/18, 89%; $\chi^2 = 2.29$, $p = 0.13$). However, nestlings received supplementary food for a longer period in 2012 (mean 10.69 ± 0.42 , range 8–12 days) than in 2011 (mean 6.23 ± 0.51 , range 4–9 days; $t = 6.54$, $df = 25$, $p < 0.0001$). Wheatears (*Oenanthe oenanthe*) and meadow pipits (*Anthus pratensis*) occasionally ‘stole’ the supplied food, but were quickly chased off by ouzels and were considered to have a negligible impact upon the amount available to ouzels. Common gulls (*Larus canus*) located the food at two territories after 3–6 days of feeding in 2011, and at six territories after 3–8 days of feeding in 2012, and rapidly emptied the trays. We then ceased the food supplementation in these territories, since no food was available for ouzels and because of the increased risk of predation of ouzel nestlings by common gulls. Such territories were subsequently removed from the experiment, since they could not be reliably categorised as either fed or control. We re-classified the six (2011) and two (2012) territories where we provided food, but never observed it being taken by ouzels, as controls. Thus, during 2011–12, we successfully provided supplementary food at 21 territories, with a further 34 territories classed as controls.

Provisioning rates

We carried out provisioning rate observations, by adults to nestlings, at 5 of 13 (38%) early, and 9 of 16 (56%) late, control territories, and at 7 of 11 (64%) early, and 8 of 9 (89%) late, fed territories in 2011. In 2012, observations were carried out at all early ($n = 16$) and late ($n = 9$) control territories, and at 12/13 (92%) early, and at all 11 late, fed territories. Supplementary food was supplied in 538/740 (73%) of deliveries to nestlings by adults at fed nests. In addition, adults were observed eating the supplementary food at 9/12 (75%) of fed territories in 2011, and at 15/17 (88%) of fed territories in 2012.

The best supported model for provisioning rate included the added positive effects of nestling age and year (Suppl. material 1: Table S1). Although the second best supported model had a $\Delta AICc$ value of 1.4, it contained one additional parameter and the same covariates as the best supported model, and was therefore not considered competitive.

Reproductive success

Reproductive success results are summarised in Table 1. There was no significant difference in early nest mean clutch size between fed and control territories in either 2011 ($t = 1.74$,

Table 1. Ring ouzel reproductive success parameters measured for early and late broods at all fed and control nests during 2011–12. Figures presented are means \pm s.e., with sample size given in parentheses. Data for mean clutch size in late nests comes only from individually identifiable colour ringed females.

Variable	2011 fed	2011 control	2012 fed	2012 control
Number of early territories	9	14	10	17
Number of late territories	7	16	8	10
Mean clutch size in early nests	4.11 \pm 0.11 (9)	3.85 \pm 0.10 (13)	4.08 \pm 0.08 (12)	3.86 \pm 0.10 (14)
Mean clutch size in late nests	5.00 \pm 0.00 (2)	4.00 \pm 0.31 (5)	4.00 \pm 0.26 (6)	4.00 \pm 0.32 (5)
Early brood fledging success	0.74 (9)	0.84 (14)	0.89 (10)	0.85 (17)
Late brood fledging success	0.74 (7)	0.87 (16)	0.87 (8)	0.70 (10)
Mean brood size at fledging in successful early nests	3.25 \pm 0.37 (8)	3.07 \pm 0.20 (14)	3.78 \pm 0.22 (9)	3.31 \pm 0.27 (16)
Mean brood size at fledging in successful late nests	4.00 \pm 0.55 (5)	3.86 \pm 0.21 (14)	3.71 \pm 0.36 (7)	3.71 \pm 0.18 (7)

$p = 0.10$) or 2012 ($t = 1.77$, $p = 0.09$), and thus no evidence of differences in territory and/or adult quality between treatment groups before supplementary feeding commenced. The top five best supported models predicting fledging success all indicated higher success for late broods compared to early broods (Suppl. material 1: Table S2). Since the brood-only model was only 1.5 Δ AICc units higher than the top-ranked model, which contained an extra parameter (year), we conclude that increased fledging success was largely associated with late broods. The top seven best supported models predicting brood size at fledging in successful nests indicated that brood size at fledging was largely determined by brood size at hatching, and the best supported model contained only this parameter (Suppl. material 1: Table S3).

BCI at fledging

Since the null model predicting individual nestling BCI at fledging received almost as much support as the two top-ranked models, we conclude that none of the models successfully predicted nestling BCI at fledging (Suppl. material 1: Table S4). The top three best supported models predicting within brood variation in nestling BCI indicated that variation was positively associated with larger broods (Suppl. material 1: Table S5). The addition of brood as a covariate, and as an interaction with brood size, added little additional support for these two models.

Juvenile post-fledging survival and return rates

The model with the highest support regarding juvenile survival for up to 100 days post-fledging indicated a positive association with BCI at fledging (Suppl. material 1: Table S6, Fig.1). The next three highest-ranked models (Δ AICc = 0.7–2.0) also contained BCI at fledging, plus between one and three additional parameters, including

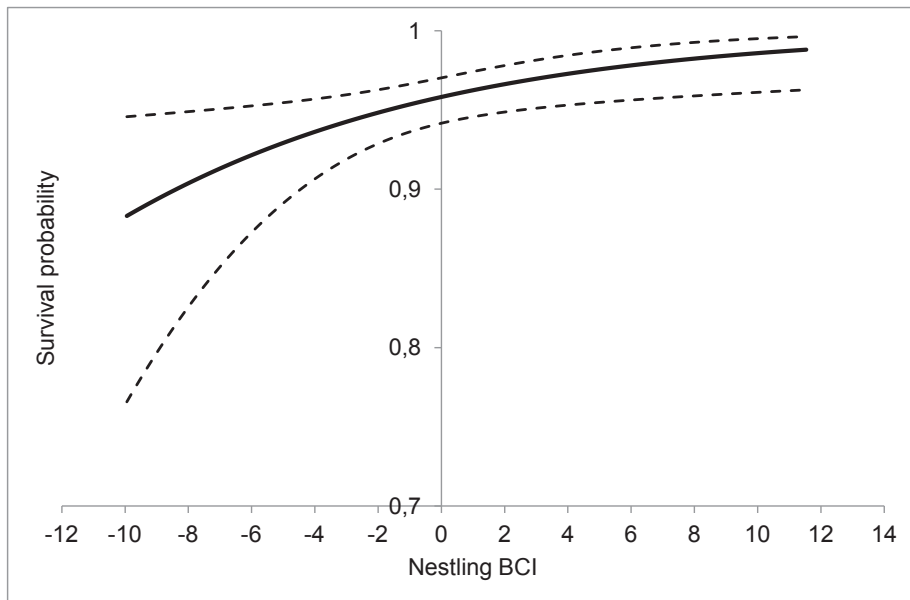


Figure 1. Predicted relationship between 4-day survival probability (solid line), \pm 95% C.I. (dashed lines) and nestling body condition index (BCI) for ring ouzels in Glen Clunie during 2011–12.

the second-ranked model which indicated a positive additive effect of the provision of supplementary food. However, since these three models all lay within 2.0 AICc values of the best supported model, and contained between one and three extra parameters, they were not considered competitive to the best supported model. There was no significant difference between treatment groups in the proportion of colour ringed nestlings that were observed returning to the study area in the years following ringing (pooling data across 2011 and 2012: control 13/152; 0.09; fed 6/117; 0.05; $\chi^2 = 1.18$, $p = 0.28$).

Causes and timing of mortality

Of the 143 juveniles fitted with transmitters, 60 (42.0%) could no longer be tracked 20–84 days after fledging and were assumed to have dispersed outside the study area, 40 (28.0%) were found dead, 35 (24.5%) shed their transmitters within the study area, and a single (0.7%) transmitter was known to have stopped working prematurely (the individual could still be identified by individual colour rings). A further seven (4.8%) transmitters were assumed to have stopped working within 16 days of fledging, since their signals became increasingly weak and intermittent in the days preceding their loss.

Of the 60 juveniles that were considered to have dispersed outside the study area, 41 (68%) had moved unusually long distances (1.5–6.0 km) from their nest sites during the 10 days preceding the estimated date at which they left the study area (nine were subsequently located outside the study area). Of the remaining 19, three were also sub-

sequently located outside the study area. This suggests that the majority of juveniles that we lost track of did in fact disperse, rather than experience transmitter failure. Of the 40 found dead, 16 (40.0%) and 11 (27.5%) were eaten and presumably killed by raptors and mammals, respectively. A further 12 (30.0%) apparently died of starvation/exposure (six deaths occurred during a 2-day period of exceptionally cold, wet and windy weather in late May 2011), and a single (2.5%) bird was apparently killed by a car.

Thirteen of the 16 (81%) deaths attributed to raptors could not be attributed to a specific species. However, the remains of single juveniles and/or their radio transmitters were found in, or within 100 m of, peregrine (*Falco peregrinus*), kestrel (*F. tinnunculus*) and sparrowhawk (*Accipiter nisus*) nests, respectively, strongly suggesting that these were the predators. None of the 11 deaths presumed to have been caused by mammals could be attributed to a specific species. However, a minimum of eight (73%) were likely killed by mustelids (stoats and weasels), because they were found in situations inaccessible to red fox, such as small holes or deep in boulder scree.

A similar proportion of early brood (14/76, 18%) and late brood (13/67, 19%) juveniles were apparently depredated ($\chi^2 = 0.02$, $p = 0.88$). During the first four days post-fledging, signs suggested that mammals were the main predator (5/6, 83%), with raptors being the main apparent predator thereafter (15/21, 71%). Mortality apparently from exposure occurred during the first eight days post-fledging, while the individual apparently killed by a car died approximately 3 weeks post-fledging.

Discussion

During 2011–12, we successfully provided supplementary food at 21 ouzel territories, with a further 34 territories classed as controls. However, none of adult provisioning rate to nestlings, fledging success, brood size at fledging, within-brood variation and individual nestling BCI at fledging, and post-fledging survival were positively associated with the provision of supplementary food. Post-fledging survival was positively associated with BCI at fledging, but there was no apparent association between BCI and the provision of supplementary food. Post-fledging mortality was apparently mainly due to predation by raptors and mustelid mammals. We therefore conclude that, during 2011–12, food supplementation did not improve the demographic rates which had previously been identified to be crucial for improving the population growth of ouzels in the UK. However, since the current and previous studies (e.g. Sim et al. 2011) have been carried out on areas with intensive predator control, these results may only be directly relevant to areas with similar levels of predator control.

Effects of food supplementation on provisioning rates

The majority of ouzel pairs that were supplied with supplementary food found it relatively quickly, usually within a day, and used it to feed themselves and/or their nestlings

in both years. Supplied food was mainly mealworms, which have been widely used to feed a number of passerine species due to their high protein and fat content, similarity to natural invertebrate food, and availability to receive in bulk at short notice (Bernard et al. 1997; Burfield 2002; Banjo et al. 2006). In this study, ouzel provisioning rate increased with nestling age as expected, and as noted for several other passerines (e.g. Goodbred and Holmes 1996; Zanette et al. 2000; Dawson et al. 2005). However, there was no evidence that food supplementation increased adult ouzel provisioning rate to nestlings, which suggests that natural invertebrate food was abundant in both years.

Effects of food supplementation on vital rates

Contrary to our predictions, we found no positive effect of food supplementation on ouzel fledging success, or brood size at fledging. These results are consistent with 9/19 (47.4%) of published studies on fledging success in small passerines, which found no positive effect of food supplementation (Robb et al. 2008). Although the provision of supplementary food resulted in increased brood size at fledging in great spotted woodpeckers (*Dendrocopos major*), it did not in great tits (*Parus major*), and brood size actually decreased in fed great and blue tits (*P. caeruleus*) in another study (Verboven et al. 2001; Harrison et al. 2010; Smith and Smith 2013).

None of our models predicting fledgling BCI successfully out-competed the intercept only model. In particular, supplementary fed individual ouzel nestlings fledged with a similar BCI to nestlings which received no supplementary food, in contrast to previous studies which indicated a positive effect of feeding on fledgling body mass (Arcese and Smith 1988; Simons and Martin 1990; Verboven et al. 2001). In this study, we found no decrease in within-brood variation in BCI in fed, as opposed to unfed, ouzel nestlings. As far as we are aware, no other studies have tested for potential effects of reducing within-brood variation in nestling BCI through the provision of supplementary food. Both these results indicate that there was no lack of invertebrate food sources for nestling ouzels.

Juvenile ouzels that fledged with a higher BCI had higher survival through the post-fledging period, a common (e.g. Yackel-Adams et al. 2006; Vitz and Rodewald 2011), though not universal (e.g. Anders et al. 1997; Kershner et al. 2004) finding in passerines. Recent work has suggested that the use of body mass alone may in some cases be a better measure of absolute body fat than indices adjusted for structural body size (Schamber et al. 2009; Labocha and Hayes 2012). In ring ouzels the adjustment for nestling body size (wing length) improved the regression of body mass on wing length³, but only marginally (see Methods). We therefore conclude that our measure of nestling body condition was effectively a measure of body mass alone, and thus accurately reflects the key BCI parameter of nestling body fat content (Schamber et al. 2009; Labocha and Hayes 2012).

There was no positive effect of food supplementation on subsequent ouzel post-fledging survival, a result consistent with the only previous passerine studies which

have tested for this effect, in the New Zealand hihi (Armstrong et al. 2007; Walker et al. 2013). In addition, we found no positive effect of food supplementation on return rate in the year following ringing in ouzels, a result consistent with that in song sparrows (*Melospiza melodia*; Arcese and Smith 1988).

Causes and timing of mortality

Mortality rates of juvenile ouzels were considerably lower during 2011–12 than those during a similar study in the same study area during 2006–08, with most deaths apparently due to predation in both time periods (Sim et al. 2013b). Far fewer juveniles with known fates were apparently predated by raptors during 2011–12 (16/100; 16%) than during 2006–08 (42/103; 41%; $\chi^2 = 15.26$, $p < 0.0001$). However, the proportion of juveniles that were apparently predated by mammals did not differ between 2011–12 (11/100; 11%) and 2006–08 (19/103; 18%; $\chi^2 = 2.23$, $p = 0.14$). Precise identity of these apparent predators was known for only a small number of individuals, but no definite predation events by buzzards (*Buteo buteo*) were noted in 2011–12, in contrast to eight assumed predation events by buzzards in 2006–08 (Sim et al. 2013b). Similar numbers of buzzards attempted to breed in the study area in both time periods (pers. obs.), so this difference was not down to lower buzzard numbers. It seems likely that alternate buzzard prey [e.g. rabbits (*Oryctolagus cuniculus*) and voles (*Microtus* sp.)] were more abundant during 2011–12 than 2006–08, resulting in fewer ouzels being taken as prey during 2011–12. As in 2006–08, mammals predated most juveniles during the first four days post-fledging, with raptors being the main predators thereafter (Sim et al. 2013b).

Conclusions

Contrary to our predictions, the provision of supplementary food had no positive effect on adult provisioning rates to nestlings, fledging success, brood size, individual BCI at fledging, or subsequent post-fledging survival, and did not decrease within-brood variation in nestling BCI in ouzels. We therefore conclude that the supply of invertebrate food sources for nestlings was not a major limiting factor in our study area during 2011–12. However, it is possible that invertebrate food sources were unusually abundant during the two-year study period, in which case the provision of supplementary food would not be expected to create a positive effect on reproductive success or juvenile survival. Most feeding experiments, including this one, are short-term in nature and may therefore fail to detect potential positive effects on population demographics of providing supplementary food in years of natural food shortage (Robb et al. 2008). Thus, an experiment which was carried out over a longer time period, and at a larger scale, than the present one may have detected positive effects of the provision of supplementary food.

It remains entirely possible that factors acting on the migration routes and/or in the wintering grounds are important in driving observed declines in ouzel numbers

in the UK, and further work is required to investigate these (Sim et al. 2010). However, in the short-term, the provision of appropriate habitat mosaics on the breeding grounds is likely to be beneficial for the conservation of ouzel populations (Burfield 2002; Sim et al. 2013a). Further studies are required to quantify the precise mix of these habitat types, at the appropriate scale, which provide concealment from predators and access to abundant food supplies throughout the spring and summer months. The provision of such a complex habitat mosaic is thus likely to be of importance in reversing long-term ouzel population declines in the UK.

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Supplementary material I

Supplementary Information

Authors: Innes M.W. Sim, Nicholas I. Wilkinson, Davide Scridel, David Anderson, Staffan Roos

Data type: species data

Explanation note: Details of the models predicting adult ring ouzel provisioning rate to nestlings, fledging success, brood size at fledging in successful nests, nestling BCI at fledging, within-brood variation in nestling BCI at fledging, and juvenile survival probability for up to 100 days post-fledging.

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A critical review of the Mediterranean sea turtle rescue network: a web looking for a weaver

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Abstract

A key issue in conservation biology is recognizing and bridging the gap between scientific results and specific action. We examine sea turtles—charismatic yet endangered flagship species—in the Mediterranean, a sea with historically high levels of exploitation and 22 coastal nations. We take sea turtle rescue facilities as a visible measure for implemented conservation action. Our study yielded 34 confirmed sea turtle rescue centers, 8 first-aid stations, and 7 informal rescue institutions currently in operation. Juxtaposing these facilities to known sea turtle distribution and threat hotspots reveals a clear disconnect. Only 14 of the 22 coastal countries had centers, with clear gaps in the Middle East and Africa. Moreover, the information flow between centers is apparently limited. The populations of the two species nesting in the Mediterranean, the loggerhead *Caretta caretta* and the green turtle *Chelonia mydas*, are far below historical levels and face a range of anthropogenic threats at sea and on land. Sea turtle rescue centers are acknowledged to reduce mortality in bycatch hotspots, provide a wealth of scientific data, and raise public awareness. The proposal for a Mediterranean-wide rescue network as published by the Regional Activity Centre for Specially Protected Areas a decade ago has not materialized in its envisioned scope. We discuss the efficiency, gaps, and needs for a rescue network and call for establishing additional rescue centers and an accompanying common online database to connect existing centers. This would provide better information on the number and types of rescue facilities on a Mediterranean scale, improve communication between these facilities, enhance standardization of procedures, yield large-scale data on the number of treated turtles and their injuries, and thus provide valuable input for targeted conservation measures.

Keywords

Mediterranean Sea, marine turtles, conservation, rescue facilities, first aid, rehabilitation, information management, networking, awareness raising

Introduction

Conservation biology is called upon to help translate scientific knowledge into specific action. Bridging this gap has been abetted by a new era of scientific endeavor (Stachowitsch 2003, Rose et al. 2011) in which the focus and urgency of scientific work has changed. In marine biology, a considerable and increasing percentage of papers is being devoted to documenting deterioration and dysfunction at population, ecosystem and global levels, often suggesting amelioration strategies. This is particularly true in the case of sea turtles, well-known flagship species, which face major anthropogenic threats at sea and on their nesting beaches worldwide. The complex life-history and highly migratory nature of sea turtles (Hamann et al. 2010), combined with the many human impacts, make conservation challenging and transcend simple, contained management measures. All sea turtle species are listed on The IUCN Red List of Threatened Species (IUCN 2014). They are a case study of marine megafauna that are now functionally or entirely extinct in many coastal ecosystems (Jackson et al. 2001).

The Mediterranean is a historically overexploited marginal sea (Morales-Muñiz and Roselló-Izquierdo 2008). It is also polluted, for example with oil and plastic, which are particularly pertinent with regard to sea turtles (Gramentz 1988). Its multitude of coastal nations with different socio-economic and cultural backgrounds—not to mention linguistic barriers—stymies the concerted protection and conservation of its sea turtle populations (Amano and Sutherland 2013). The loggerhead *Caretta caretta* (Linnaeus, 1758) is the most common species in the Mediterranean (Broderick et al. 2002, Margaritoulis et al. 2003). Like the green turtle *Chelonia mydas* (Linnaeus, 1758), it nests in the eastern regions (Kasperek et al. 2001, Margaritoulis et al. 2003), but also frequents different habitats during different life stages in the western sector (Garofalo et al. 2013), where it co-occurs with individuals from the Atlantic (Carreras et al. 2006, Casale et al. 2008a, Wallace et al. 2010). Leatherbacks *Dermochelys coriacea* (Vandelli, 1761) are observed regularly at sea (Casale et al. 2003), whereas the hawksbill turtle *Eretmochelys imbricata* (Linnaeus, 1766) and Kemp's ridley *Lepidochelys kempii* (Garman, 1880) are recorded here only occasionally (Laurent and Lescure 1994, Camiñas 2003). Sea turtles have been afforded protection under a number of treaties and laws (Suppl. material 1). The EU Habitats Directive lists all five species in Annex IV as being of community interest and in need of strict protection (European Community 1992). It also protects the loggerhead and green turtle as priority species with the need for special conservation areas (Annex II). Accordingly, these two species are subject to wide-ranging scientific research, monitoring, and conservation efforts by dozens of universities and institutions, along with numerous dedicated associations and societies, e.g., ARCHELON, EuroTurtle, MEDASSET.

Fisheries bycatch, boat strikes, intentional killing, and entanglement in marine debris including ghost gear have been identified as the main threats at sea (Tomás et al. 2008, Casale and Margaritoulis 2010, Casale et al. 2010, Casale 2011). On land, degradation and reduction of nesting habitat caused by touristic and recreational

activities, light pollution, noise, construction, sand extraction, and traffic (Camiñas 2004) are taking their toll. Turkey, for example, holds the largest green turtle nesting population in the Mediterranean with about 230 females nesting annually (Seminoff 2004). Only three generations back, from 1879–1919, about 3,500 individuals still nested there (Seminoff 2004). The Turkish population has experienced a 93% decline over the last 95 years. Green turtles face a measurable risk of extinction worldwide and therefore qualify for the IUCN Endangered status under Criteria A2bd (Seminoff 2004). The Mediterranean population is genetically distinct from Atlantic populations and belongs to a separate regional management unit (RMU) (Wallace et al. 2010), which is recognized to face high threats and a high risk of extinction (Wallace et al. 2011). A total of 339–369 females are nesting in the entire Mediterranean (Broderick et al. 2002).

The resident Mediterranean loggerhead population is genetically isolated (Carreras et al. 2011) from individuals of the two Atlantic RMUs migrating to Mediterranean foraging grounds and is considered to face high threats (Wallace et al. 2011). Bycatch rates are estimated at up to 200,000 loggerheads per year, leading to more than 50,000 deaths annually through direct interaction alone (Casale 2008, Lucchetti and Sala 2010, Casale 2011). Considerable declines on specific nesting beaches have been recorded (Ilgaz et al. 2007). Overall, no significant population trend could be observed on nesting beaches over the last decades, but survival probabilities are somewhat lower than would be expected from a healthy population (Casale et al. 2014). Anecdotal information suggests a decline over decadal scales (Casale and Margaritoulis 2010). The latest IUCN assessment classified the loggerhead as vulnerable (Marine Turtle Specialist Group 1996) but did not specifically deal with the Mediterranean population and needs updating (IUCN 2014). Total population estimates are not available, neither are total stock mortality estimates (Camiñas 2004).

The importance of dedicated rescue facilities for sea turtles was recognized during the 1980s (RAC/SPA 2004). One of the first rescue centers in the Mediterranean was established by ARCHELON in Greece in 1994 (Suppl. material 1). Others followed, but not all of them provided full facilities and treatment (Bentivegna 2005). After two decades of ill-concerted development, the need for regulating and improving sea turtle rescue was generally acknowledged, and guidelines for the standardization of rescue activities were established. The Regional Activity Centre for Specially Protected Areas (RAC/SPA) published a rough outline of a Mediterranean-wide sea turtle rescue network. It should consist of sea turtle rescue centers and first-aid stations (hereafter referred to as STRCs and FASTs, respectively) strategically based along the coasts, each adhering to common established guidelines of conduct (RAC/SPA 2004). RAC/SPA proposed to potentially act as the network's international coordinator (see Suppl. material 1 for more details).

The most obvious function of STRCs and FASTs is the rescue and rehabilitation of individual turtles. STRCs, though “in the last line of defense”, are a management tool that acts on a number of fronts. Firstly, they help increase adult and subadult survival rates, a major priority in conservation action (Camiñas 2004) that has a

considerable effect at the population level. STRCs play a significant role in reducing indirect mortality of stranded, injured or comatose adults, and large juveniles. These are size classes typically affected by the two major threats at sea: bycatch and ship strikes. Adult survival is the main factor in population growth rates; large juveniles have a high reproductive value (Wallace et al. 2008). Modeling approaches show that population persistence is much more sensitive to the survival of older age classes than to that of the first year cohort, i.e., eggs and hatchlings (Chaloupka 2002, Mazaris et al. 2005, Mazaris et al. 2006). Secondly, research is an acknowledged function of STRCs (RAC/SPA 2004). STRCs are often linked to stranding network activities. They provide a wealth of scientific data (Shaver and Teas 1999, Casale et al. 2010) on each stranded or floating turtle, including the nature of their injuries. Stranding records represent a valuable source of information on both mortality factors and spatio-temporal distribution (Casale et al. 2010). Thirdly, awareness campaigns targeting fishers to reduce post-release mortality of bycaught turtles are an urgent priority mitigation measure (Casale et al. 2007a, Echwikhi et al. 2012, Domènech et al. 2014). RAC/SPA (2004) acknowledged the importance of STRCs in sensitizing fishers to the plight of sea turtles through awareness programs, handling workshops, and promoted cooperation between fishers and conservation biologists. Thusly trained fishers are more likely to bring in comatose turtles for medical treatment before releasing them back into the sea (Casale et al. 2007a, Domènech et al. 2014). They are also more likely to adopt the simple onboard handling procedures (Gerosa and Aureggi 2001) that can considerably reduce indirect mortality of bycaught turtles. Fourthly, public awareness campaigns also target non-professionals, who may visit STRCs and partake in educational activities or volunteer. STRCs therefore bridge the gap between science and everyday life. They can get people involved in the conservation cause and potentially initiate a public conservation movement (Casale et al. 2007a).

Despite these efforts, clear gaps remain in the protection of Mediterranean sea turtles. In trawl fisheries, for example, proven management strategies such as turtle excluder devices (TEDs) are not routinely employed (Laurent et al. 1996, Casale et al. 2004), but their future use in EU waters is currently being discussed. Outdated fishing gear is apparently sold to non-EU countries, perpetuating illegal fishing practices in both larger-scale and artisanal fisheries (Lucchetti and Sala 2010). Bycatch in the latter, small-scale fisheries can even exceed levels in industrial fisheries (Lewison et al. 2014) and is particularly relevant in the Mediterranean (Echwikhi et al. 2012). This adds to the urgency of a functioning and tight-knit rescue network and supranational awareness programs.

We take the number of rescue facilities as a visible measure for practical progress in sea turtle conservation and provide an update on currently operating facilities (STRCs, FASTs, informal or temporary institutions) with verified contact details and information on the history of each center (Table 1). We then examine whether the coverage of rescue facilities correlates with threat hotspots and the distribution, including nesting sites, of the two sea turtle species nesting in the Mediterranean, and whether the

Table 1. Sea turtle rescue facilities in the Mediterranean. Based on literature search, available internet data, and personal communications. All listed websites were last accessed on 8 Feb. 2015, with final updates April 2015. All given contact details were confirmed by staff members of the respective rescue facilities, unless stated otherwise. Encompasses 20 countries bordering the Mediterranean, along with the islands of Cyprus and Malta (Northern Cyprus: self-declared state; Gibraltar: British Overseas Territory; Ceuta and Melilla: Spanish autonomous cities on the N. African coast). Countries and rescue facilities listed in alphabetical order. Italian rescue facilities ordered alphabetically by administrative region. Names of contact persons without academic titles. FAST, first-aid station; FI, figure icon: corresponding icon in Figs 1–2; IRF, informal or temporary rescue facility; NP, National Park; STRC, sea turtle rescue center.

Country	FI	Type	Contact details & additional information
Albania, ALB	!	STRC	Planned for 2014: establishment of a STRC in Patok, NW-Albania, within the framework of the IPA Adriatic CBC Programme, co-funded by the EU (Sajmir Beqiraj, University of Tirana, personal communication, 8 Feb. 2014).
Algeria, DZA			No data available.
Bosnia–Herzegovina, BIH		—	No sea turtle rescue facilities (Tarik Kupusović, Hydro-Engineering Institute Sarajevo, personal communication, 24 July 2013).
Croatia, HRV	1	STRC	Blue World Institute of Marine Research and Conservation, Sea Turtle Rescue Center, Lošinj Marine Education Centre, Kaštel 24, 51551 Veli Lošinj, www.blue-world.org (section “News/July 2013”), Contact person: Jelena Basta (Education director), Email: info@blue-world.org , jelena.basta@blue-world.org , Tel.: (+385) 51–604666, Cell: (+385) 91–6046667. Opened on 19 July 2013. Part of NETCET (Network for the Conservation of Cetaceans and Sea Turtles in the Adriatic, co-funded by the EU IPA Adriatic CBC Programme), http://www.netcet.eu/ .
	2	STRC	Marine Educational Centre Pula, Marine Turtle Rescue Centre, Aquarium Pula, Fort Verudela, Verudela bb, 52105 Pula, http://www.aquarium.hr/ , Contact person: Karin Gobić Medica, Email: infos@aquarium.hr , karin@aquarium.hr , Tel.: (+385) 52–381402, Cell: (+385) 91–1381414. Part of NETCET; http://www.netcet.eu/ .
Cyprus, CYP	3	STRC	CyMARC, Cyprus Marine Aquaculture Research Center, c/o Ministry of Agriculture, Natural Resources and Environment, Marine Environment Division, Department of Fisheries & Marine Research (DFMR), 1416 Nicosia, http://www.moa.gov.cy/moa/dfmr/dfmr.nsf/DMLAqa_en/DMLAqa_en , Contact persons: George Anastasiades (Responsible scientist), Email: director@dfmr.moa.gov.cy , Tel.: (+357) 24–422888, Marina Argyrou (Senior Fisheries and Marine Research Officer), Email: margyrou@dfmr.moa.gov.cy , Tel.: (+357) 22–807852. Located at Meneou, Larnaca; formerly known as Meneou Marine Research Station (MeMARS).
Northern Cyprus, CTR		—	No sea turtle rescue facilities (Wayne Fuller, European University of Lefke, Society for the Protection of Turtles (SPOT), Marine Turtle Conservation Project (MTCP), personal communication, 25 Jan. 2014).
Egypt, EGY	!	IRF	Monitoring takes place every year; veterinarians take care of injured and stranded sea turtles in the field (Moustafa Fouda, Ministry of State for Environmental Affairs, Nature Conservation Sector, personal communication, 21 Jan. 2014).
France, FRA	4	STRC	Centre d'Etudes et de Sauvegarde des Tortues Marines de Méditerranée, Avenue du Palais de la Mer, BP 106, 30240 Le Grau-du-Roi, www.cestmed.org , Contact person: Jeanbaptiste Senegas, Email: contact@cestmed.org , Tel.: (+33) 4–66515737, Cell: (+33) 6–24475155.

Country	FI	Type	Contact details & additional information
Corsica	!	IRF	Local network dedicated to sea turtle research and rescue, http://www.corsenetinfos.fr/Un-reseau-d-alerte-tortues-marines-en-Corse_a10556.html . Part of the French Mediterranean Sea Turtle Network (Réseau Tortues Marines de Méditerranée Française, RTMMF; http://lashf.fr/laSHF/commissions/RTMMF). Ongoing collaboration with the nearby Sardinian STRC of Asinara (see Nr. 29) (Michel-Jean Delaugerre, personal communication, 21 April 2015).
	!	IRF	Helping Hand Trust, 10, Queensway Quay, http://www.helpinghand.gi/ , Tel.: (+350) 200–73719, Emergency numbers: (+350) 200–72500, 199 (Police), or (+350) 200–65964 (Department of the Environment). Calls will be redirected to the rescue team. The organization takes care of sea turtles together with veterinarians from the Gibraltar Veterinary Clinic and representatives of the Department of the Environment. (Stephen Warr, H.M. Government of Gibraltar, Department of the Environment, personal communication, 21 Jan. 2014).
Greece, GIB	5	EAST	ARCHELON, Sea Turtle Protection Society of Greece (STPS), First Aid Station, Amvrakikos Bay (Koprina), http://www.archelon.gr/ , Contact person: Pavlos Tsaros, Email: rescue@archelon.gr , Cell: (+30) 697–2877962.
	6	EAST	ARCHELON, Sea Turtle Protection Society of Greece (STPS), First Aid Station, Pagalohori of Rethymno, Arkadi, Crete, http://www.archelon.gr/ , Contact person: Reggina Stefanatou, Email: archelon.crete@gmail.com , Cell: (+30) 693–7352379.
	7	STRC	ARCHELON, Sea Turtle Protection Society of Greece (STPS), Sea Turtle Rescue Centre, 3 rd Marina, Glyfada 16675, Athens, http://www.archelon.gr/ , Contact person: Pavlos Tsaros, Email: rescue@archelon.gr , Tel.: (+30) 21–08944444, Cell: (+30) 694–151151.
	8	STRC	Hellenic Centre for Marine Research (HCMR), Hydrobiological Station of Rhodes, Sea Turtle Rescue Centre, Cos Street, 85100 Rhodes, www.hcmr.gr , Contact person: Maria Corsini-Foka, Email: hmr@hmr-nmr.gr , Tel.: (+30) 2241–027308, (+30) 2241–078320.
Israel, ISR	9	STRC	Israel Sea Turtle Rescue and Rehabilitation Center, Mevo'ot Yam, Mikhmoret, http://old.parks.org.il/ , Contact person: Yaniv Levy (Sea turtle project coordinator, STRC-Director, ISRAEL National Nature and Parks Authority), Email: Israelseaturtle@npa.org.il , yaniv@npa.org.il , Tel.: (+972) 9–8669173, Emergency number: *6911 (Speed dial service).
Italy, ITA			National emergency numbers, toll free: 1530 (National Coast Guard), 800–904841 (Europ Assistance Service “SOS tartarughe”, http://www.europassistance.it/azienda/progetto-tartanet/). Calls will be redirected to the nearest rescue center. (Daniela Casprini, Associazione Vittime della Caccia, personal communication, 21 July 2013).
Basilicata	10	STRC	Centro di recupero tartarughe marine di Policoro, Via Lido, 75025 Policoro (MT), Email: policoro@leganavale.it , Tel.: (+39) 0835–403814, Cell: (+39) 335–1272335, Contact person: Raffaele Micelli (Educational director), Cell: (+39) 335–1272336. Founded by the sailing club “Circolo Velico Lucano” in 1999, http://www.circolovelicolucano.it (news postings). Located in the Nature Reserve “Bosco Pantano”. http://www.tartanet.it/ .
	11	EAST	Oasi WWF Policoro Herakleia, Riserva Regionale “Bosco Pantano”, C.R.A.S. (Wildlife Rescue Center), Piazza Siris, Località Idrovora, 75025 Policoro (MT), http://www.oasiwwfpolicoro.net/ , Contact person: Antonello Palmisano, Email: wvf.poli@gmail.com , Tel.: (+39) 0835–1825157. The center is part of WWF Italy's Sea Turtle Network and provides first aid through a team of veterinarians and marine biologists. Temporary holding tanks are available. Turtles in need of surgery are transferred to the Veterinary Hospital of Bari.
Calabria	12	STRC	Centro Recupero Tartarughe Marine Brancalene, Piazza Stazione, Brancalene Marina, 89036 Brancalene (RC), http://www.naturalmentebrancalene.org/ , Email: naturalmentebrancalene@gmail.com , Cell: (+39) 328–3020921, (+39) 340–1290736. (Simona Clò, formerly in charge of the now partly defunct rescue network “Tartanet”, personal communication, 27 Jan. 2014).

Country	FI	Type	Contact details & additional information
Campania			Sea Turtle Rescue and Rehabilitation Center, Area Marina Protetta "Capo Rizzuto", Centro Direzionale, Via C. Colombo, 88900 Crotone (KR), http://www.ampcaporizzuto.it/ (news postings), Email: segreteria@ampcaporizzuto.it , Tel.: (+39) 0962-665254, Contact person (administrative): Simone Scalise, Email: scalise@ampcaporizzuto.it , Contact persons (technical & scientific): Pierfrancesco Cappa, Domenico Piro (Veterinarian). The MPA has joined the National Action Plan for the Conservation of Sea Turtles (PATMA). It is collaborating with WWF Calabria, and the Universities of Calabria, Pisa, and Bari. The STRC was funded by the state, region, province, and the EU. It is run by MPA staff and through the help of volunteers. Turtles are rehabilitated in the Aquarium in Capo Rizzuto, where CEAM, the Environmental Education Center, pursues public information activities.
	13	STRC	
	14	STRC	Centro di Recupero Tartarughe Marine di Punta Campanella, Via Padre Rocco 40, 80061 Massa Lubrense (NA), Email: cea@puntacampanella.org , Tel.: (+39) 081-8089877. http://www.tartanet.it/ .
	15	STRC	Sea Turtle Rescue Centre, Stazione Zoologica Anton Dohrn, Villa Comunale, 80121 Napoli, http://www.szn.it/SZNNWeb/showpage/115?languageId_2=2 , Contact person: Sandra Hochscheid, Email: sandra.hochscheid@szn.it , Tel.: (+39) 081-5833222.
Emilia-Romagna	16	STRC	Turtle Point, Sea Turtle Rescue and Rehabilitation Centre, Stazione Zoologica Anton Dohrn, Via Cocchia 28, 80124 Bagnoli (NA), http://www.szn.it/SZNNWeb/showpage/115?languageId_2=2 , Contact person: Sandra Hochscheid, Email: aquarium@szn.it , sandra.hochscheid@szn.it , Tel.: (+39) 081-7629338. Specialized in rehabilitation phase after treatment and prior to reintroduction into the wild.
	17	STRC	ARCHE', Via Mulinetto 40/a, 44100 Ferrara (FE), Email: archeturtle@tiscali.it , Tel.: (+39) 0532-767852. (Daniela Casprini, Associazione Vittime della Caccia, personal communication, 17 July 2013).
	18	STRC	Fondazione Cetacea Onlus, Viale Torino 7/A, 47838 Riccione (RN), http://fondazionecetacea.org/ , Contact person: Valeria Angelini, Email: informazione@fondazionecetacea.org , educazione@fondazionecetacea.org , Tel.: (+39) 0541-691557. Part of NETCET, http://www.netcet.eu/ .
Friuli-Venezia Giulia	19	EAST	WWF Area Marina Protetta di Miramare, Cetacean and Sea Turtle Monitoring and First Aid Group, Viale Miramare 349, 34151 Grignano-Trieste (TS), http://www.riservamarinamiramarare.it/ (news postings), Contact person: Francesco Zuppa, Email: info@riservamarinamiramarare.it , zuppa@riservamarinamiramarare.it , Tel.: (+39) 040-224147. Part of WWF Italy's Sea Turtle Network.
	20	STRC	Centro cura tartarughe marine, Università degli Studi di Bari Aldo Moro, Facoltà di Medicina Veterinaria, Piazza Umberto I 1, 70121 Bari (BA), http://www.uniba.it/ricerca/dipartimenti/dipmedveterinaria/strutture/centro-cura-tartarughe-marine/ , Emergency number, toll free: 800-883046. (Daniela Casprini, Associazione Vittime della Caccia, personal communication, 21 July 2013).
Puglia	21	STRC	Centro Recupero Tartarughe Marine del Salento, Parco Naturale Regionale Bosco e Paludi di Raucio, 73100 Lecce (LE), http://www.salento.com/il-salento/varie/centro-recupero-tartarughe-marine-crtm-del-salento . (Chiara Caputo, CRTM Museo di Calimera, personal communication, 30 July 2013). http://www.tartanet.it/ .
	22	STRC	Centro Recupero Tartarughe Marine Manfredonia-Legambiente, Parco Nazionale del Gargano, Oasi Lago Salso, 71043 Manfredonia (FG). (Simona Clò, formerly in charge of the now partly defunct rescue network "Tartanet", personal communication, 27 Jan. 2014).
	23	STRC	CRTM Museo di Calimera, SP 275 Calimera-Borgagne, km 1, 73021 Calimera (LE), http://www.msns.it/ , Contact person: Chiara Caputo, Email: osservatorio.faunistico@msns.it , chiara.caputo@msns.it , Cell: (+39) 324-8898814, (+39) 320-6586558.

Country	FI	Type	Contact details & additional information
Sardegna			Public emergency number, toll free: 1515 (Sardinian Forest Service). In addition to national emergency numbers, see above. The nearest rescue team will be alerted. All of the following Sardinian rescue facilities operate within the Sardinian Regional Network for the Conservation of Marine Turtles and Mammals.
	24	FAST	Area Marina Protetta di Tavolara Punta Coda Cavallo, Via Dante 1, 07026 Olbia (OT), www.amptivolara.com , Contact person: Pier Panzalis, Email: ambiente@amptivolara.it , Office/Emergency Tel.: (+39) 0789–203013. Turtles in need of veterinary treatment will be transferred to CReS in Oristano (see Nr. 25).
	25	STRC	Area Marina Protetta “Penisola del Sinis—Isola di Mal di Ventre”, Centro di Recupero del Sinis delle tartarughe marine e dei cetacei (CReS), Office: Piazza Eleonora 1, 09072 Cabras (OR), Email: ambiente@areamarinasinis.it , Office Tel.: (+39) 0783–391097. Rescue center located at IAMC-CNR (National Research Council), Loc. Sa Mardini, 09170 Oristano (OR), http://www.areamarinasinis.it/ , Contact person (administrative): Giorgio Massaro, Email: direzione@areamarinasinis.it , Cell (Emergency): (+39) 340–1096633, Contact person (technical & scientific): G. Andrea de Lucia (Scientific coordinator), Email: a.delucia@iamc.cnr.it , Cell (Emergency): (+39) 339–4654779.
	26	FAST	Centro di Primo Soccorso “Capo Carbonara”, Via degli Asparagi 51/a, 09049 Villasimius (CA), http://www.ampeapocarbonara.it/pagina.php?id=48 , Contact persons (Biologists): Francesca Frau, M. Francesca Cinti, Fabrizio Atzori, Email: info@ampeapocarbonara.it , Tel.: (+39) 070–790234, Cell (Emergency): (+39) 320–4643038. The center, formerly a “node” of the regional rescue network, is presently in phase of acknowledgement as an official FAST by the Ministry of the Environment. Expected new address: Via degli Oleandri 6/b, 09049 Villasimius (CA).
	27	STRC	Centro Recupero Cetacei e Tartarughe marine “Laguna di Nora”, Centro di educazione ambientale Laguna di Nora, Laguna di Nora Loc. Nora, 09010 Pula (CA), http://www.lagunadinora.it/sezione.php?idsez=5 , Contact person: Giuseppe Ollano, Email: info@lagunadinora.it , gollano@lagunadinora.it , Tel.: (+39) 070–9209544. The center is managed on behalf of the Municipality of Pula.
Sicilia	28	FAST	First Aid Station, National Park “Arcipelago di la Maddalena”, http://www.lamaddalena-park.it/ (news postings), Email (NP Environmental Office): ufficio.ambiente@lamaddalena-park.org . The FAST is located in the park’s Environmental Education Center in Stagnali on Isola Caprera. Contact person: Yuri Donno, Tel.: (+39) 0789–790233. For 2014, regional funding was expected, and a new operational plan, regarding the coordination of wildlife rescue operations through a specialized company, was being devised. The NP will continue giving first aid and providing public information services.
	29	STRC	Parco Nazionale dell’Asinara, Area Marina Protetta “Isola dell’Asinara”, Centro Recupero Animali Marini, Cala Reale, Isola dell’Asinara, 07046 Porto Torres (SS), www.parcواسينارا.org , http://www.cramasinara.org/ , General information: parco@asinara.org , enteparcoasinara@pec.it , Emergency contact: Associazione CRAMA, which manages the rescue center on behalf of the NP. Email: info@cramasinara.org , Cell: (+39) 340–8161772.
	!	STRC	Centro Recupero Provinciale Fauna Selvatica e Tartarughe Marine, SP 29, 92011 Catolica Eraclea (AG), Contact person: Calogero Lentini (Veterinarian), Email: aldolentini@alice.it , Additional information: http://www.tartanet.it/ . The center was closed in Dec 2013. Reopening is planned but not scheduled yet. (Simona Clò, formerly in charge of the now partly defunct rescue network “Tartanet”, personal communication, 27 Jan. 2014).

Country	FI	Type	Contact details & additional information
	30	STRC	Centro Recupero Tartarughe Marine di Linoisa, Via Pozzolana di Ponente 13, 92010 Linoisa (AG), http://www.marineturtle.it/ , Contact person: Stefano Nannarelli (Director), Email: Info@marineturtle.it , Tel.: (+39) 0922-972076.
	31	STRC	Centro Soccorso e Cura Tartarughe Marine, WWF Italia, Stazione Marittima 92010 Lampedusa (AG), http://www.lampedusaturtle.org/ , http://www.lampedusaturtle.org/ , Tel.: (+39) 338-2198533.
	32	STRC	DELPHIS Aeolian Dolphin Center, Via Simone Neri 1, 98050 S. Marina Salina, Isola di Salina, Isole Eolie (ME), http://www.delphisad.c.it/ , Contact person: Renata Mangano (Coordinator), Email: delphiscenter@gmail.com , Cell (Emergency): (+39) 333-1932002. DELPHIS takes care of sea turtles and cetaceans in the Aeolian Islands.
	33	FAST	Filicudi Wildlife Conservation, Pronto Soccorso Tartarughe Marine dell' Arcipelago Eoliano, Office: Località Stimpagnato, Isola di Filicudi, 98055 Lipari (ME), http://www.filicudiconservation.com/ , Email: info@filicudiconservation.com , Cell: (+39) 349-440201. The FAST is located at Hotel Phenicusa, Via Porto 7, 98050 Filicudi, Isole Eolie (ME). The Visitor Information Center is located in Piazzetta Pecorini Mare, Filicudi (ME). (Sources: http://www.filicudiconservation.com/ , http://www.oraresortphenicusa.com/ .)
Toscana	34 35	STRC	NECTON Marine Research Society, Office for East Sicily: Via Guido Gozzano 47, 95100 Catania; Office for West Sicily: Via Celona 11, 98165 Ganzirri Messina; http://www.necton.it/ , Contact person: Renata Mangano (Coordinator), Email: info@necton.it , Cell (Emergency): (+39) 333-1932002.
	!	IRF	Acquario di Livorno, Piazzale Mascagni 1, 57127 Livorno (LI), http://www.acquariodilivorno.it/ , Email: info@acquariodilivorno.it , Tel.: (+39) 0586-269111/154 (in case of emergency, ask for the Aquarium Department). The Aquarium takes care of injured and stranded sea turtles along the coast of Tuscany, together with Costa Edutainment's veterinarians and specialists. It provides holding tanks for treatment and rehabilitation. The ultimate aim is the animals' reintroduction into the wild.
	36	STRC	Centro didattico WWF dei Ronchi, Centro di educazione ambientale e Centro recupero tartarughe marine, Via Donizetti, Località Ronchi, 54038 Marina Di Massa (MS), http://centrodidatticowfronchi.wordpress.com/ , Contact person: Gianluca Giannelli (Director), Email: parcodidattico@virgilio.it , Cell: (+39) 360-234789. Reopened on 3 August 2013. (Paolo Casale, Scientific coordinator of the Sea Turtle Project WWF Italy, personal communication, 29 Jan. 2014).
			No data available.
Lebanon, LBN			Marine Biology Research Centre (MBRC) Tajura, PO Box: 30830 Tajura (Casale and Margaritoulis 2010, http://www.ciism.org/online/institutes/inst/Inst142.htm). Status unknown.
Libya, LBY	?	STRC	
Malta, MLT	!	IRF	Nature Trust Malta (www.naturetrustmalta.org) runs a temporary rehabilitation center located at Malta Aquaculture Research Centre, San Lucjan Tower, Marsaxlokk. A permanent center is planned at Xrobb I-Ghagin in Dellimara. Contact persons: Vincent Attard, Karen Goode, Responsible veterinarian: Anthony Grupetta, Email: info@naturetrustmalta.org , Cell (Emergency): (+356) 9999-9505.
Monaco, MCO	37	STRC	Musée océanographique, Institut océanographique, Fondation Albert 1er, Prince de Monaco, Av. Saint-Martin, MC 98000 Monaco, http://www.oceano.mc/en/activities/activities-for-all/-/the-museum-involved-in-helping-turtles , Contact person: Pierre Gilles (Head of Aquarium), Email: p.gilles@oceano.mc , Tel.: (+377) 93153646. Part of the French Mediterranean Sea Turtle Network (Réseau Tortues Marines de Méditerranée Française, RTMMF).

Country	FI	Type	Contact details & additional information
Montenegro, MNE		—	No sea turtle rescue facilities (Milena Baraković, Environmental Protection Agency of Montenegro, Department for nature protection, monitoring, analysis and reporting, personal communication, 22 July 2013).
Morocco, MAR	!	IRF	ATOMIM (Association de protection des Tortues Marines au Maroc), Department of Biology, Faculty of Science, PO Box 2121, Tétouan 93002 Morocco, http://www.atomm.org/ , Contact person: Mustapha Aksissou (Director), Email: aksissou@yahoo.fr , Cell: (+212) 661–953689.
Palestine, State of, PSE			No data available.
Slovenia, SVN	!	IRF	Aquarium Piran, Kidričevo nabrežje 4, 6330 Piran–Pirano, Tel.: (+386) 5–1602554, Contact person: Valter Žiža, Email: akvarij.piran@guest.arnes.si , Cell: (+386) 41–975386; Golob d.o.o., Zatočišče za živali prosto živečih vrst, Glavni trg 7, 2366 Muta, Tel.: (+386) 2–8761285, Contact person: Zlatko Golob, Cell: (+386) 41–518939. Veterinarians of the Wildlife Sanctuary “Zatočišče za živali prosto živečih vrst” take care of injured sea turtles. Aquarium Piran provides space for first aid treatment; it does not, however, have holding tanks for a longer rehabilitation phase.
Spain, ESP	38	STRC	Centro de Recuperación de Animales Marinos de la Fundación GRAM, Passeig de la Platja 28–30, 08820 El Prat de Llobregat (Barcelona), http://cram.org/ , Contact person: Elsa Jiménez, Email: info@cram.org , vet@cram.org , Tel.: (+34) 937–524581.
	39	STRC	Centro de Recuperación de Especies Marinas Amenazadas, CREMA, de Málaga, Calle Pacífico 80, 29003 Málaga, http://www.auladelmar.info/crema , Contact person: José Luis Mons Checa, Email: crema@auladelmar.info , Tel.: (+34) 952–229287, Emergency number: 112.
	40	STRC	Centro de Recuperación de Fauna La Granja de El Saler, Av de los Pinares 106, 46012 El Saler, Valencia, Contact person: Juan Eymar, Email: centre_granja@gva.es , Tel.: (+34) 96–961610847. Local government property.
Ceuta			No data available.
Melilla			No data available.
Syria, SYR		—	No sea turtle rescue facilities (Alan F. Rees, IUCN Marine Turtle Specialist Group (MTSG) Regional Vice Chair for the Middle East Region, personal communication, 24 Jan. 2014).
Tunisia, TUN	41	STRC	National Institute for the Sciences and Technologies of the Sea (INSTM), Station de Protection et de Soin des Tortues Marines Monastir, Route de Khniss, 5000 Monastir, B.P. 59, Tel.: (+216) 73–531867, Contact persons: Kaouthar Maatouk, Email: maatoukk@yahoo.fr , Olfa Chaieb, Email: olfa.chaieb@instm.mrt.tn . (Imed Jribi, University of Sfax, Faculty of Sciences, personal communication, 24 Jan./7 Feb. 2014).
Turkey, TUR	42	STRC	DEKAMER, Sea Turtle Research, Rescue and Rehabilitation Centre, Dalyan, Muğla, http://caretta.pamukkale.edu.tr/ , Contact person: Yakup Kaska, Email: caretta@pau.edu.tr , dekamer@pau.edu.tr , Tel.: (+90) 252–2890077, Cell: (+90) 533–5753339. Affiliated with Pamukkale University.

number of facilities seems adequate. We take fisheries bycatch—the most important source of anthropogenic sea turtle mortality at sea—as a proxy for threats (Lutcavage et al. 1997, Casale 2008, Casale 2011).

Methods

Sea turtle rescue facilities currently in operation in the Mediterranean

Kasperek's (2001) and RAC/SPA's (2004) earlier lists of STRCs served as a starting point for our compilation. Casale and Margaritoulis (2010) and RAC/SPA's List of Focal Points for SPAs provided useful information on potential contact persons and their email addresses. We searched for Mediterranean rescue facilities (STRCs and FASTs) on the internet between August 2012 and March 2014 using Google Search and Duck Duck Go. Searching for phrases such as “list of Mediterranean sea turtle rescue centers”, “first aid stations for marine turtles”, “sea turtle rescue”, and respective variants in English, French, Italian, and Spanish yielded abundant information. Much, however, was incomplete, outdated, and difficult to substantiate, i.e., gray literature, websites without “date of latest update”, or PDFs without mastheads.

The next step involved obtaining full contact details by sending inquiries directly to the centers. When basic contact information was lacking or no data were available for a particular country, we sent inquiries to official institutions, i.e., ministries of the environment, animal welfare organizations, universities, and national park administrations. We also asked the rescue center contacts about other facilities in their vicinity.

Sea turtle distribution, threat hotspots, and coverage with rescue facilities

We reviewed the peer-reviewed literature to identify the key distribution areas, i.e., nesting beaches, feeding and overwintering areas, foraging sites of juveniles, and major migration corridors, of loggerhead and green turtles.

We also reviewed the peer-reviewed literature to identify threat hotspots for sea turtles in the Mediterranean. Of the commonly acknowledged main threats, i.e., nesting habitat degradation, bycatch, ship strikes, and direct exploitation, we chose fisheries bycatch as a proxy for threats because: 1) it affects primarily older individuals and has great impact on population levels; 2) bycatch and its geographic distribution are quantifiable; 3) it occurs Mediterranean-wide and year-round; 4) its effects can be mitigated by STRCs. We briefly discuss the main deployment areas of the three critical fishing gear types, i.e., trawl, drifting longline, set nets (Lucchetti and Sala 2010).

To better visualize the geographic coverage of rescue facilities and its appropriateness, we compare current locations of rescue facilities to 1) key sea turtle distribution areas (Fig. 1), 2) bycatch hotspots in the three crucial gear types (Fig. 2), and 3) RAC/SPA's (2004) proposed rescue network (Fig. 2).

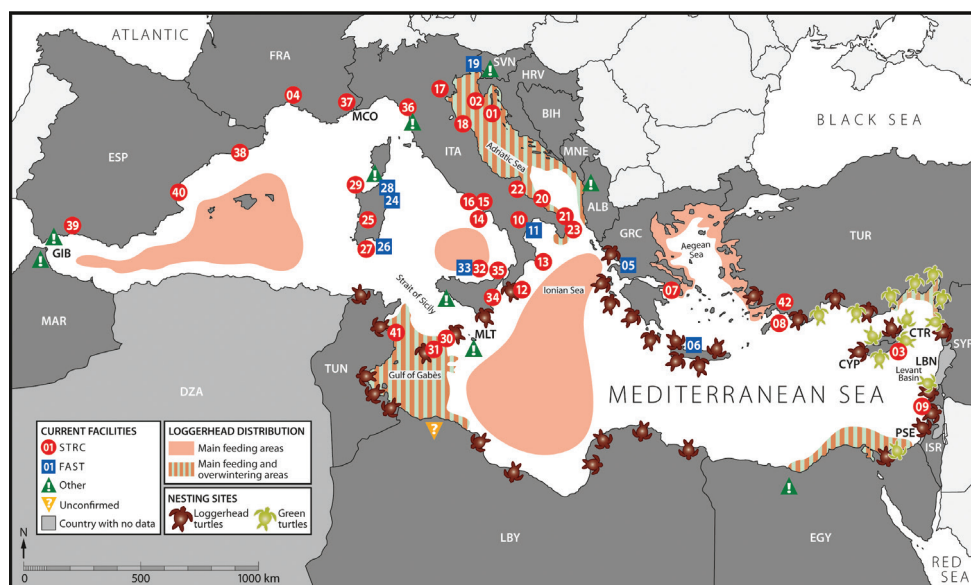


Figure 1. Current sea turtle rescue facilities in relation to nesting sites and distributional hotspots. Locations of rescue facilities are based on available internet data and personal communications; latest update: April 2015. Sea turtle distribution and nesting sites combined and modified in part after Gerosa and Casale (1999), Kasperek et al. (2001), Margaritoulis et al. (2003), Mingozi et al. (2007), Casale and Margaritoulis (2010), Echwiki et al. (2012), Schofield et al. (2013), Stokes et al. (2015). For country abbreviations, contact details, and additional information on facilities see Table 1. FAST, First Aid Station; STRC, Sea Turtle Rescue Center.

Results and review of current status

Sea turtle rescue facilities currently in operation in the Mediterranean

The literature and internet search for rescue facilities yielded 34 STRCs, eight FASTs, and seven informal or temporary rescue institutions (Table 1, Figs 1–2). Based on website content and personal communications, we assumed that the listed STRCs and FASTs adhere to RAC/SPA's code of conduct.

Italy held 21 STRCs, six FASTs, and one informal rescue facility. Spain had three STRCs, Croatia and Greece each had two STRCs, the latter also featuring two FASTs. France held one STRC and a local rescue network in Corsica. Cyprus, Israel, Monaco, Tunisia, and Turkey held one STRC each. Malta held a temporary rehabilitation center. While Egypt, Gibraltar, Morocco, and Slovenia did not have any formal rescue facilities, injured turtles were cared for by veterinarians, local animal welfare organizations and rescue associations. Albania, Bosnia and Herzegovina, Montenegro, Northern Cyprus, and Syria had no facilities. Data for Libya were available but could not be verified. For Algeria, Lebanon, and the State of Palestine, no data on rescue facilities were found and no official institutions reached.

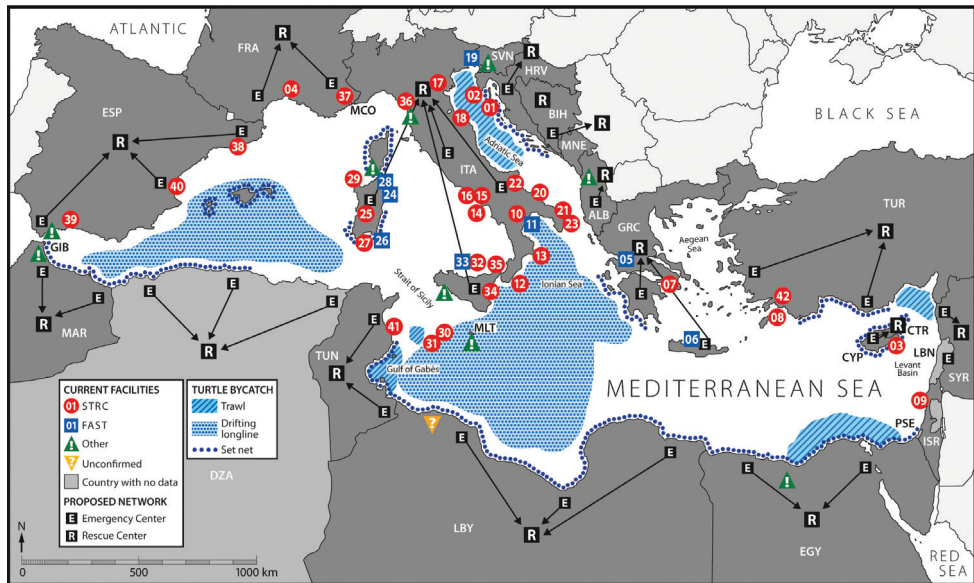


Figure 2. Current sea turtle rescue facilities in relation to the proposed network and bycatch hotspots. Locations of rescue facilities are based on available internet data and personal communications; latest update: April 2015. Proposed network after RAC/SPA (2004). Bycatch areas combined and modified in part after Luccetti and Sala (2010) and Echwikh et al. (2012). Detailed geographic data on set net bycatch not available for all shorelines. For country abbreviations, contact details, and additional information on facilities see Table 1. FAST, First Aid Station; STRC, Sea Turtle Rescue Center.

Sea turtle distribution

Ninety-nine percent of green turtle nesting takes place in Turkey and Cyprus (Kasperek et al. 2001) (Fig. 1). The five most important nesting beaches are located in the east of Turkey close to Syria, and on the northern coast of Cyprus. Minor nesting activity also occurs in Syria, Lebanon, Israel, and Egypt (Camiñas 2004, Rees et al. 2008, Stokes et al. 2015) (Fig. 1). Pelagic developmental and neritic habitats were commonly thought to be restricted to the eastern Mediterranean (Levant basin) (Camiñas 2004). Recently, satellite-tagged individuals revealed an important migratory corridor from Turkey and Cyprus to Egypt and Libya, with two major foraging hotspots in the latter country (Stokes et al. 2015).

Loggerhead reproductive habitats and main foraging grounds are concentrated in the wider eastern basin (Casale and Margaritoulis 2010) (Fig. 1). Main nesting sites are located in Greece, Turkey, Cyprus, and Libya but nesting also occurs in Tunisia, Israel, Syria, Lebanon, Italy (Mingozzi et al. 2007), and Egypt (Margaritoulis et al. 2003, Camiñas 2004) (Fig. 1). Loggerheads frequent different habitats during different life stages. As juveniles grow, they are increasingly found in neritic habitats, switching from epipelagic prey caught in oceanic areas to benthic prey (Schroeder et al. 2003, Casale et al. 2008b). Adults show extended fidelity to their neritic feeding

grounds (Broderick et al. 2007). Individuals of the Mediterranean RMU (Wallace et al. 2010) range throughout the basin; Atlantic transients visit foraging habitats in the western Mediterranean.

Continental shelves and slopes constitute the main adult loggerhead feeding areas (Fig. 1). The northern Adriatic is an important foraging area (Casale et al. 2010) for adults from Greek (Lazar et al. 2004, Carreras et al. 2006, Giovannotti et al. 2010, Garofalo et al. 2013) and Turkish rookeries, as well as for juveniles (Casale et al. 2004). Other important areas are the southern Adriatic (Casale et al. 2012a), the Ionian Sea, the Strait of Sicily, and the Tunisian shelf (Margaritoulis et al. 2003, Casale et al. 2007b, Casale et al. 2014). The latter is frequented by turtles from Greek rookeries, Libya, and the Atlantic (Garofalo et al. 2013).

In the central Mediterranean between Italy, Tunisia, and Libya, potential neritic and pelagic foraging habitats are close to each other (Casale et al. 2008b) (Fig. 1). The Pelagic Islands, for example, are a core foraging ground here (Casale et al. 2012b). Individuals from the key Mediterranean turtle rookeries, i.e., western Greece, Crete, and Cyprus, frequent these central Mediterranean feeding grounds (Margaritoulis et al. 2003, Broderick et al. 2007, Casale et al. 2008a, Zbinden et al. 2008). Other important neritic foraging areas are off southeastern Turkey, the Egyptian coast (Gerosa and Casale 1999), and the Spanish coast (Gómez de Segura et al. 2006).

The Gulf of Gabès (Tunisia, Libya) and the northern Adriatic with their wide shelves are also well-known resting and wintering areas (Camiñas 2004) (Fig. 1). Moreover, the southern Adriatic and to a lesser extent the northern Adriatic and Ionian sea are important developmental areas for loggerheads in the first four years of their lives (Casale et al. 2009). Preferred loggerhead habitats off the Spanish Mediterranean coast are characterized by a large number of juveniles, especially around the Balearic islands during spring and summer (Camiñas and de la Serna 1995) and around Columbretes Island Marine Reserve throughout the year (Gómez de Segura et al. 2003).

The North African coast is apparently an important migratory pathway for loggerheads across the Mediterranean (Broderick et al. 2007, Casale et al. 2012b). Seasonal movements include southbound emigration movements from the northern Adriatic (Lazar et al. 2003, Zbinden et al. 2008, Zbinden et al. 2011) and movements of juveniles in the western Mediterranean (Cardona et al. 2009). The central Mediterranean, in particular the Strait of Sicily, is probably a key route for turtles migrating between the eastern and the western Mediterranean basins (Casale et al. 2007a).

Threat hotspots (fisheries bycatch)

Casale (2008) estimated the incidental captures of Mediterranean sea turtles by fishing gear at 150,000 per year and the associated mortality at 50,000. The respective values in a more recent analysis remained in the same high range, namely 132,000 and 44,000 (Casale 2011). Camiñas (2004) identified the Spanish and Italian surface longline fisheries, northern Adriatic Italian trawl, Tunisian trawl, Turkish trawl, Mo-

roccan driftnet, and Italian driftnet fisheries as having the highest impact (Fig. 2). The Adriatic and Ionian Seas and the Strait of Sicily show high bycatches by trawlers (Casale et al. 2004, 2007a) and longliners (Deflorio et al. 2005, Casale et al. 2007a) (Fig. 2). The central Mediterranean was recognized as a bycatch hotspot of pelagic longline and bottom trawl fisheries (Jribi et al. 2007, 2008, Casale et al. 2007a, Echwikhi et al. 2010, 2012; Fig. 2). Artisanal fisheries are the most important in terms of numbers of boats and fishermen involved (Camiñas 2004, Cambiè 2011). Accordingly, Casale (2011) identified small-scale fisheries (versus large vessels) as a key concern and pinpointed the eastern basin as a hotspot.

Discussion

Nature conservation and species protection have developed in direct response to the deterioration of many habitats and the decline of many species. Nonetheless, there is a continuing gap between problem recognition in the form of scientific data and specific management actions. Moreover, the process is typically reactive rather than proactive. We proffer that this gap should be the narrowest for endangered flagship species in habitats that are well-defined and have a long history of scientific research. Sea turtles in the Mediterranean are a case in point. Just as the analysis of sea turtle protection coverage (based on nesting sites) revealed clear gaps on a global level (Mazaris et al. 2014), our review (based on rescue center coverage) revealed clear gaps on the Mediterranean level as well.

Compared to the international conventions and protocols of the 1970s that first recognized the problems facing sea turtles in the Mediterranean, most STRCs and FASTs were founded quite late (Suppl. material 1). Moreover, the quality criteria for these facilities were formulated even later (RAC/SPA 2004).

Compared to Kasparek's (2001) and RAC/SPA's (2004) preliminary lists of about a dozen rescue centers, the number of known rescue facilities in the Mediterranean has quadrupled within the last ten years. With nearly 50 facilities today (Figs 1–2), the dimension of RAC/SPA's (2004) proposed rescue network has now been reached. Nonetheless, there are major differences between the proposed network and the current situation, specifically in type and distribution.

Regarding type, RAC/SPA envisaged a network consisting of 16 rescue centers, each connected to between two and four emergency centers (Fig. 2). Currently, however, there are about four times as many STRCs as FASTs. The original proposal might have been based on more FASTs as a more cost-effective solution. One potential explanation for the current proportion, though, is that FASTs proved to be suboptimal because most injured turtles require long-term veterinary treatment in a rescue center. Alternatively, our web-based search might have missed FASTs because they probably have smaller budgets and are less well represented on the internet. This is an invitation to complete and regularly update our list of Mediterranean sea turtle rescue facilities.

Regarding distribution, RAC/SPA's (2004) proposal for an even spread along the Mediterranean coasts has not been realized. For example, whereas the original pro-

posal indicated 17 facilities along the southern Mediterranean coastline (5 STRCs, 12 FASTs), only 3 are currently confirmed. The distribution along the northern coast more closely resembles RAC/SPA's proposal. Once Albania's planned STRC (Table 1) is in effect, the same will hold true for the eastern Adriatic. Nonetheless, some regions (Italy) exhibit numerous closely adjoining STRCs, while others have long stretches without a single facility.

More than half (28) of all facilities are concentrated in Italy. This is the best fit because the total estimated bycatch there (23,600 = 18%; Casale 2011) is the highest of any Mediterranean country. Spain, with the second highest bycatch (20,920 = 16%) is covered by only 3 facilities. Nonetheless, Spain shows the best agreement in facility number and location with RAC/SPA's proposal (Fig. 2). The third, fourth and fifth highest bycatches are reported for Tunisia (17,600 = 13%), Morocco (15,400 = 12%), and Turkey (12,900 = 10%). Here the correlation is extremely poor, each country having only a single facility. This is particularly evident in the case of Turkey, which hosts many key nesting beaches but has only one very recently established STRC. Overall, there is a pronounced shortage of facilities along the Middle East and African Mediterranean coasts, especially when considering that Casale (2011) identified the North African continental shelves, the Levantine basin and the Aegean Sea as the areas in which sea turtles were most at risk from bottom trawling, demersal longlines, and set nets.

The deficits in STRC numbers and distribution no doubt partially reflect the different cultures, socio-economic status, funding priorities and perception of environmental issues of the 22 Mediterranean countries. It is further compounded by security issues in many countries (Amano and Sutherland 2013). Along with language barriers, these factors are also a hindrance to the originally envisioned concept of a functioning network. In many cases there seems to be limited knowledge about and communication between neighboring centers—within and across borders. This was detected at the national level, e.g., in the 15 Italian administrative regions with access to the sea, and even at local level, i.e., two facilities in one town. Clearly, knowledge of neighboring institutions and communication between centers are key ingredients for successful collaboration on the national level, which in turn is a prerequisite for rewarding international cooperation.

What are the repercussions for the current status of Mediterranean sea turtles? Primarily, suboptimal species protection. Injured individuals with chances of survival are not being detected and not receiving the necessary veterinary care. Each adult turtle is thought to represent one surviving individual out of an estimated 500–1000 hatchlings that emerge from their nests on the beach. This, coupled with the life history of sea turtles—slow growth, long period before sexual maturity—means that every adult is very important. Considering the historical decline in sea turtle numbers and the present low numbers, the mortalities must be reduced. We must avoid the situation—known for cetaceans such as the vaquita in the Gulf of California, the Western Pacific gray whale, or the North Atlantic right whale—that further human-induced mortalities of any individual, in particular adult females, will jeopardize species or population survival (Kraus et al. 2005). This is precisely where the role of STRCs grasps: every rescued individual counts.

What is the vision for the future and how can this be achieved? The goal must be to work toward a dense and evenly distributed rescue facility network. Communication between centers must be improved. This will not only help rescue individual sea turtles but will provide added benefits—in a positive feedback loop—for the other STRC functions, namely research and public outreach (RAC/SPA 2004). What species and life stages are being treated and what injuries are being suffered in what regions? The actual overall numbers of treated and saved individuals—related to the number of nesting females—is the first important data set that STRCs could deliver. STRCs are eminently suited to provide answers, which are currently not available and which are not necessarily supplied by today's hypothesis-driven scientific literature (Casale and Margaritoulis 2010). Such information, compiled Mediterranean-wide and collated by a coordinating level, can then be translated into improved management. The relatively small size of the Mediterranean may actually help in collecting such information: many critical areas are located within Exclusive Economic Zones, simplifying data collection and promoting a sense of responsibility.

Such data could best be compiled using a common online database. This is a viable option based on our experience at the level of individual STRCs: most contacted rescue center staff were readily willing to help, seemed interested in information dissemination, and eager for news about other facilities. We therefore support setting up a "Mediterranean Sea Turtle Rescue Network Database" online, containing and updating all the basic information on rescue facilities (Table 1). The idea of an internet gate or special webpage for communication, containing publicly available profiles of existing rescue facilities, is not new (Kasperek 2003, Panagopoulou and Rees 2009, also see Suppl. material 1). The online database could be hosted by recognized pan-Mediterranean institutions, e.g., RAC/SPA or EuroTurtle, ensuring a wide audience from the onset. It would be an inexpensive yet effective tool for adequate coordination and monitoring of rescue and conservation efforts. Beyond facilitating communication among sea turtle specialists, marine scientists and non-professionals, it would help standardize data collection and presentation.

General conclusions

Our review shows that sea turtle rescue facilities—as visible and measureable evidence of concrete conservation action—are characterized by a:

- a) relatively late start in light of early conventions and protocols addressing sea turtle threats,
- b) relatively late set of quality criteria,
- c) slow increase in number,
- d) patchy distribution with major unserved regions,
- e) often haphazard rather than problem-oriented correlation between sea turtle distribution/threat hotspots and rescue facility sites,

- f) still insufficient number,
- g) poor readily available information on and suboptimal communication between these facilities.

Information on the number of treated turtles and their injuries is essential input for further, well-targeted, and concerted conservation measures. At the same time, we must go beyond the often heard call for more data and apply common sense to the ongoing threatened status of sea turtles in the Mediterranean and elsewhere. Further action should not be delayed until further evidence has been collected. A functioning network of sea turtle rescue centers would be a good first step in this direction.

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Supplementary material I

Stepping stones toward a Mediterranean sea turtle rescue network

Authors: Judith Ullmann, Michael Stachowitsch

Data type: species data

Explanation note: Historical outline of sea turtle protection in the Mediterranean.

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Status and management of non-native plant invasion in three of the largest national parks in the United States

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Abstract

Globally, invasion by non-native plants threatens resources that nature reserves are designated to protect. We assessed the status of non-native plant invasion on 1,662, 0.1-ha plots in Death Valley National Park, Mojave National Preserve, and Lake Mead National Recreation Area. These parks comprise 2.5 million ha, 23% of the national park land in the contiguous USA. At least one non-native species inhabited 82% of plots. Thirty-one percent of plots contained one non-native species, 30% two, 17% three, and 4% four to ten non-native species. Red brome (*Bromus rubens*), an ‘ecosystem engineer’ that alters fire regimes, was most widespread, infesting 60% of plots. By identifying frequency of species through this assessment, early detection and treatment can target infrequent species or minimally invaded sites, while containment strategies could focus on established invaders. We further compared two existing systems for prioritizing species for management and found that a third of species on plots had no rankings available. Moreover, rankings did not always agree between ranking systems for species that were ranked. Presence of multiple non-native species complicates treatment, and while we found that 40% of plots contained both forb and grass invaders, exploiting accelerated phenology of non-natives (compared to native annuals) might help manage multi-species invasions. Large sizes of these parks and scale of invasion are formidable challenges for management. Yet, precisely because of their size, these reserves represent opportunities to conserve large landscapes of native species by managing non-native plant invasions.

Keywords

Exotic species, invasibility, invasive plants, multiple species, prioritization, elevation

Introduction

Non-native species are those transported intentionally or unintentionally by human activities to new areas (typically new continents) outside of their long-term evolutionary habitat (Schulz and Gray 2013). Nature reserves world-wide contain increasing numbers of non-native plant species in their flora (e.g., Allen et al. 2009, Barnett et al. 2007, Pyšek et al. 2003). Invasion by non-native plants occurred both before and after establishment of nature reserves. Even the world's oldest national park, Yellowstone established in 1872 in the USA, contained at least one non-native plant species in the initial 1886 inventory of the park (Whipple 2001). It now contains over 180 non-native species. At least two non-native plant species inhabited Kruger National Park in South Africa when the park was first designated in 1898, and it now contains at least 257 non-native plant species (Foxcroft et al. 2008). Non-native species comprise 15% of the flora in Czech Republic nature reserves (Pyšek et al. 2002), 6% in Australia's Kakadu National Park (Cowie and Werner 1993), 11% in Villarrica National Park in Chile (Pauchard and Alaback 2004), and 13% in Gros Morne National Park in Canada (Rose and Harmanutz 2004).

If invading species were all innocuous and simply added to a reserve's biodiversity, there might be little cause for concern (Schlaepfer et al. 2011). Unfortunately, numerous examples exist of harmful non-native plants that can 'engineer' new habitat by creating/destroying ecosystem structure or modifying natural disturbance regimes, devastating native species (Foxcroft et al. 2013). Several observations portend reason for concern with ongoing invasion of nature reserves. Lag times – delays of decades to centuries between introduction of a species and exponential population growth – suggest that some currently sparse and innocuous species could become invasive (Theoharides and Dukes 2007). Moreover, short-term persistence of native species in invaded reserves might mask extirpations as eventual outcomes, indicating only part of the full impact of invasions is yet evident (Gilbert and Levine 2013). Plant invasions also can impact native biota in ways difficult to readily observe, such as inducing genetic change in native species (Leger 2008). On one hand, adaptation by native species to co-existing with a non-native might help natives persist, but on the other hand, genetic fitness for adaptation to other stressors (e.g., climate change) could be compromised. A precautionary approach would not assume that ongoing invasion will be innocuous (Andreu and Vilà 2010).

While the difficult task to curtail undesired species introductions between continents and into nature reserves needs further attention (Lodge et al. 2006), existing plant invasions are not necessarily hopeless situations due to some key aspects of invasion ecology. Existence of 'lag times' means that a relatively broad time window can be available for treating invaders before exponential population growth begins and

treatment becomes arduous (Theoharides and Dukes 2007). Because high elevations of reserves are often least invaded, they represent locations managers can prioritize to keep relatively free of non-natives (Pauchard et al. 2009).

Identifying the non-native species present and their distribution is a first step in managing biological invasions (Barnett et al. 2007). For example, inventorying species abundance enables identifying which species are currently rare and thus manageable through early detection and treatment (Klinger et al. 2006). Requiring species inventory data, prioritization systems are under development in many areas and rank species for treatment based on factors such as species traits, impacts, and management difficulty (Andreu and Vilà 2010, Randall et al. 2008). Identifying sites invaded by multiple species is important, because invaders differing in phenology can necessitate multiple treatments at different times (Marushia et al. 2010).

Here, we collected and analyzed a unique data set of non-native plant species in three of the four largest national parks in the contiguous USA. The survey totaled 2.5 million ha, 8% of the total land area managed by the National Park Service and 23% of the USA's national park land outside of Alaska. Using a plot-based approach to assess over 1,600 sites, we examined the following questions: (1) How many non-native plant species were detected and what were the most and least frequent species among parks? (2) How similar was non-native plant species composition among parks? (3) Were species prioritization rankings similar between ranking systems and related to relative abundance of species? (4) How many sites contained multiple non-native species, and which species co-occurred? (5) Were species distributions associated with elevation gradients and how similar were distributions among parks? Findings have implications for species distribution mapping, design of early detection and monitoring, and formulating non-native plant management plans for nature reserves.

Methods

Study areas

We conducted the study in three parks managed by the U.S. National Park Service: Death Valley National Park, Mojave National Preserve, and Lake Mead National Recreation Area, in the U.S. states of California, Nevada, and Arizona (Fig. 1). Each park exceeds 0.5 million ha and includes the largest national park in the lower 48 states (Death Valley), the third largest (Mojave), and the fourth largest (Lake Mead). These parks are in the Mojave Desert, where landforms include canyons, alluvial fans, cinder cones, low hills, mountains, dry lake beds, and intermittently flowing stream channels (Fig. 2). Predominant vegetation types include desert holly (*Atriplex hymenelytra*) and other shrub communities in the lowest-elevation basins, creosote bush-bursage (*Larrea tridentata*-*Ambrosia dumosa*) shrubland to elevations of 1200 m, blackbrush (*Coleogyne ramosissima*) shrubland at middle elevations, single-leaf pinyon-Utah juniper (*Pinus monophylla*-*Juniperus osteosperma*) woodland starting around 1,600 m on mountain

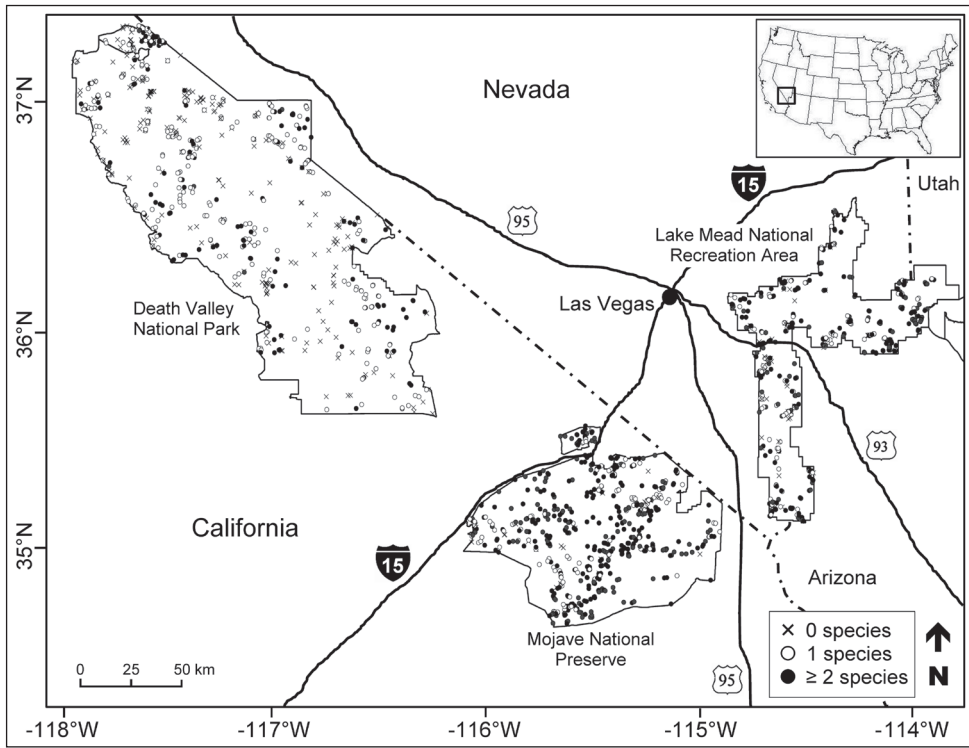


Figure 1. Location of three parks managed by the National Park Service in which we measured non-native plant species on 1,662 plots, Mojave Desert, southwestern USA.

slopes, and conifer forest such as white fir (*Abies concolor*) or bristlecone pine (*Pinus longaeva*) on the highest peaks (Keeler-Wolf 2007). Most annual plants are winter annuals, germinating in fall/winter (beginning in November) and growing until April (Beatley 1974).

Among the parks, Death Valley contains the lowest (along the Death Valley floor) and highest elevations (Telescope Peak in the Panamint Mountains; Table 1). Mojave Preserve contains low-lying basins and its highest elevations in the Clark Mountains, with much of the park of intermediate elevation (800–1500 m). Spirit Mountain, at 1,720 m in the Newberry Mountains, is the highest peak in Lake Mead National Recreation Area. Climate varies across the region and with elevation. The Death Valley, California, weather station at 58 m below sea level receives only 6 cm/yr of precipitation and has an average January daily low temperature of 4 °C and July high of 47 °C (1961–2012 records; Western Regional Climate Center, Reno, Nevada). In contrast, a station 1,326 m in elevation receives 27 cm/yr of precipitation, with a January average daily low of 3 °C and July high of 34 °C (1958–2011; Mitchell Caverns, California, in south-central Mojave Preserve).

Before they were designated, the parks incurred anthropogenic disturbance including clearing for townsites, agriculture, or ranches in the 1800s and early 1900s;



Figure 2. Views of national parks showing the variety of contexts in which non-native plants occur. Death Valley National Park: top: Death Valley floor where non-natives were generally sparse; middle: an area previously dominated by native shrubland and converted largely to non-native *Bromus* annual grassland following wildfire; bottom: Panamint Mountains where *Bromus tectorum* was the major non-native species. Mojave National Preserve: top: developed area with a history of human occupation and disturbance (Zzyzx, California); middle: *Yucca brevifolia*-*Coleogyne ramosissima* mature native shrubland, among the most susceptible communities to wildfire spread facilitated by non-native grasses; bottom: this community type following wildfire. Lake Mead National Recreation Area: top: *Tamarix* spp. (tall, green, leafy trees) infesting riparian areas around the Lake Mead shoreline; middle: shoreline activities can distribute non-native plants, making treating non-natives along the shoreline a priority for park managers; bottom: natural washes can serve as vectors for dispersal of non-natives.

localized mining; alteration to springs and seeps (e.g., piping water elsewhere); road and trail building; and ranching operations with cattle and sheep (Lovich and Bainbridge 1999). Non-native burros were kept as work animals by miners and continue to inhabit these parks as feral animals (Beever and Pyke 2005). Livestock grazing allotments were decommissioned in the late 1990s in Lake Mead National Recreation Area and partially decommissioned from 1998–2002 in Mojave National Preserve. Extensive roadless areas exist, but the parks do contain widespread road networks, such as 3,700 km of roads within Mojave National Preserve (Vogel and Hughson 2009). Combined human visitation to the parks was 7.8 million visitors in 2012, including 1 million in Death Valley, 0.5 million in Mojave, and 6.3 million in Lake Mead (National Park Service, Public Use Statistics Office, Denver, Colorado).

Table 1. Characteristics of parks and sample plots for assessing non-native species distribution in National Park Service lands in the Mojave Desert, USA.

	Death Valley	Mojave	Lake Mead National
	National Park	National Preserve	Recreation Area
Park characteristics			
Size (ha)	1,345,321	643,112	563,513
Elevation range (m)	-86 to 3,368	270 to 2,417	158 to 1,720
Sample plots			
No. plots	623	600	493
Plot elevation range (m)	-86 to 3,329	276 to 2,416	158 to 1,704
Plots with ≥ 1 non-native species (%)	65	95	78
Plots with > 1 non-native species (%)	22	73	55
Plots with > 2 non-native species (%)	3	32	28
Maximum non-native species/plot	10	6	10
Non-native species/plot (mean ± SEM)	0.92±0.03	2.05±0.04	1.96±0.06
Total non-native species on plots	22	17	22
Park non-native species lists			
Total non-native species	83	73	74

Stratification for sampling

We sampled all three parks using similar stratified-random designs. We divided Death Valley National Park into 16 zones corresponding to major mountain ranges (e.g., Panamint Mountains) or valleys (e.g., Death Valley floor). Using an existing vegetation map of the park (5-ha minimum mapping unit; Thomas et al. 2004) and a Geographic Information System random point generator (ArcGIS 9.3, Esri Corp., Redlands, California), we generated 5 potential points for sampling within each vegetation type. For instance, the largest zone (Last Chance Mountain Range) contained 43 vegetation types and had 215 potential sample points. Sample points were then evaluated and field visited in random order within zones, with the goal of sampling 2 (the first 2, if possible) of 5 potential sites. Potential points were rejected because of safety concerns (e.g., cliff faces were not sampled) or unsuitability (e.g., developed areas such as campgrounds), and the next potential point was evaluated.

We divided Mojave National Preserve into 31 zones according to broad landforms (e.g., Cima Volcanic Field) in a 1:100,000-scale geologic map (Miller et al. 1991). Then, to capture elevational variation, each zone was stratified by 250-m elevation bands (e.g., 750–1,000 m). Finally, these elevation bands were stratified by predicted land cover based on the Thomas et al. (2004) vegetation map. Again using a random point generator, we selected three points for potential sampling within each vegetation type × elevation × landform zone stratum. Generation of three potential sample points within a stratum provided field crews the ability to reject unsuitable sites, as at Death Valley.

The Thomas et al. (2004) vegetation map did not extend to Lake Mead National Recreation Area, so the park was stratified into 1-km² pixels based on climate (derived

from PRISM), topography (digital elevation models), and soil parent material (Lato 2006). Climate layers included July average maximum temperature ($> 41^{\circ}\text{C}$, $\leq 41^{\circ}\text{C}$), January average minimum temperature ($> 2.5^{\circ}\text{C}$, $\leq 2.5^{\circ}\text{C}$), average annual May through October precipitation ($> 8\text{ cm}$, $\leq 8\text{ cm}$), and average annual November through April precipitation (6–9 cm, 9–11 cm, and 11–16 cm). Topography was categorized as drainage, flat ($< 1\%$ slope gradient), gently sloping (1 to $\leq 10\%$ gradient), and for slope gradients $> 10\%$, by slope aspect as northeast ($0\text{--}89^{\circ}$), southwest ($180\text{--}269^{\circ}$), or neutral ($90\text{--}179^{\circ}$ or $270\text{--}359^{\circ}$). There were 188 unique combinations of these variables extant on the landscape, and 2–3 points were randomly sampled per combination.

Data collection

We used the same procedures for field data collection in all three parks. At each sample point, we surveyed a square plot of 0.1 ha for areal cover of non-native plant species (including annual, biennial, and perennial plants) using the following cover classes: present but $< 1\%$, $1\text{--}5\%$, $> 5\text{--}15\%$, $> 15\text{--}25\%$, $> 25\text{--}50\%$, $> 50\text{--}75\%$, and $> 75\%$. We recorded both live and dead annual plants as a measure of cumulative presence for two reasons: 1) live annual plants are ephemeral, absent many years and when present, for only a short time in winter/spring; and 2) fuel provided by dead annual plants poses a fire hazard to mature Mojave Desert plant communities (Brisbin et al. 2013). The length of time that dead annuals persist as upright stalks varies, but Beatley (1966) noted that red brome (*Bromus rubens*), a major non-native in the Mojave Desert, can stand approximately two years. We summed cover of live and dead stalks of annual plants into a single cover estimate by species for each plot. Nomenclature and classification of species by longevity/growth forms (e.g., perennial forb) and native/exotic followed Natural Resources Conservation Service (2013).

We established a total of 1,662 plots that encompassed 99% of elevation ranges within parks (Table 1). We sampled high elevations in warmer months and low elevations in cooler months. We worked in Death Valley National Park between May 17 and July 2, 2010, and between January 4 and May 22, 2011. Between September 30, 2010 and June 7, 2011, we sampled Mojave National Preserve. We sampled Lake Mead National Recreation Area between February 18 and May 13, 2010, and between September 9 and October 22, 2010.

Data analysis

For all plots combined and each park separately, we calculated the total number of non-native species, percentage of plots containing one or more non-native species, mean non-native richness (species/0.1-ha plot), and frequency of each species. We used Pearson correlation coefficients to examine relationships between elevation and non-native species richness and cover. We compared species prioritization rankings from

two systems: NatureServe's I-rank (Randall et al. 2008) and the California Invasive Plant Council system (Cal-IPC; Warner et al. 2003). To compare species recorded on plots with all known records of non-native species within each park, we obtained species lists maintained by each park. Using PC-ORD v. 6 (McCune and Mefford 1999), we calculated Sørensen similarities between parks of species composition (presence/absence data) recorded on plots and from park species lists.

Results

All plots across parks

Eighty-two percent of plots contained at least one non-native plant species (Table 1, 2). Non-native richness ranged from 0–10 species/0.1 ha, with a median of 2 species and mean of 1.60 ± 0.03 (\pm standard error of mean). Thirty-one percent of plots contained one non-native species, 30% two, 17% three, and 4% four to ten species. For example, considering two species of *Bromus*, 36% of plots contained neither species, 3% only cheatgrass (*Bromus tectorum*), 50% only *Bromus rubens*, and 10% both (Fig. 3). Total non-native cover ranged from 0–81%, with a median of 0.4% and mean of $2.5 \pm 0.1\%$. Most plots had low cover and few had high cover: 60% had < 1% cover, 26% had 1–5%, 8% had 5–10%, and 6% had > 10% cover.

Of 29 total non-native species on plots, 59% were annuals, 10% annuals/biennials, 14% annual to perennials, and 17% perennials (Table 2). By growth form, 55% were forbs, 35% grasses, 7% shrubs, and 3% trees. Annual forbs (31%) and annual grasses (28%) were the most prevalent groups. The most frequent species included: *Bromus rubens* (60% of plots), redstem filaree (*Erodium cicutarium*; 39%), *Schismus* spp. (28%), *Bromus tectorum* (13%), prickly Russian thistle (*Salsola tragus*; 4%), Sahara mustard (*Brassica tournefortii*; 4%), and saltcedar (*Tamarix ramosissima*; 3%).

Results were mixed regarding availability of species prioritization rankings, consistency between ranking systems, and relationship between a species' rank and its frequency (Table 2). Nineteen of 28 taxa (68%, with *Schismus* spp. grouped to genus) had rankings available from the Cal-IPC system and 17 were ranked by the NatureServe I-rank system. There were six species (21%) not ranked by either. Of the 9 taxa ranked by both systems, consistency varied. Ranking was consistent for *Bromus tectorum* and *Tamarix ramosissima*, with both systems ranking the species as high priority and capable of pervasive impacts. However, the two systems returned opposite rankings for *Schismus* spp. Cal-IPC ranked the taxon as 'low' priority, while NatureServe ranked it as 'high' priority.

We did not detect an overall correlation between elevation and non-native richness or cover (Fig. 4). The only trend apparent was that all plots containing > 2 non-native species occurred at elevations < 2,000 m. Individual species displayed stronger relationships with elevation than did total non-native measures (Fig. 5). *Brassica tournefortii* and *Schismus* spp. were most frequent at elevations below 1,200 m. Although infesting

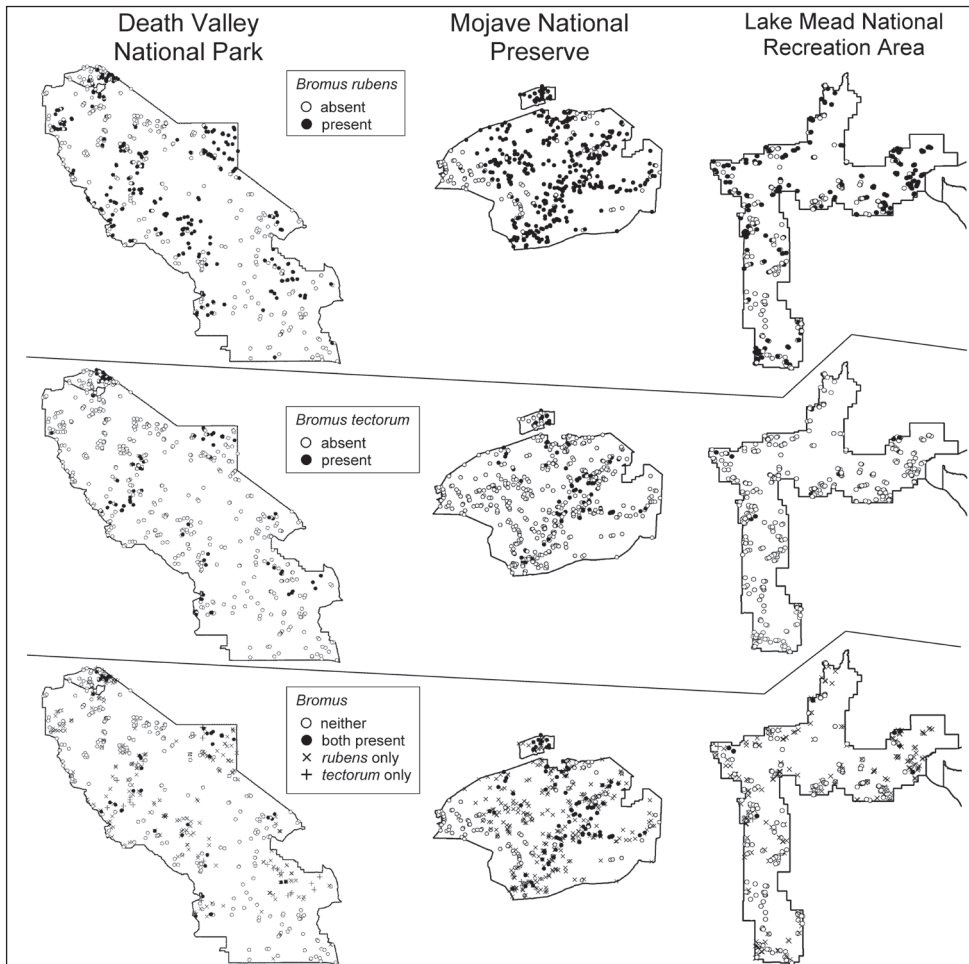


Figure 3. Presence or absence of the non-native annuals *Bromus rubens* and *Bromus tectorum* in national park units of the Mojave Desert, southwestern USA.

a broad elevation range, *Erodium cicutarium* was most frequent at middle elevations between 400 and 1,600 m. Distribution of the most frequent species, *Bromus rubens*, also was centered on middle elevations. *Bromus tectorum* exhibited a different pattern: it was most frequent at elevations above 1,600 m in Death Valley National Park and Mojave National Preserve, and at the highest elevations present in Lake Mead National Recreation Area.

Comparison of parks

The total number of non-native species detected on plots within parks was similar, ranging from 17–22 species/park (Table 1). However, mean non-native richness/plot was

Table 2. Summary of 29 non-native plant species detected within 1,662 sample plots spanning three National Park Service units in the Mojave Desert, USA.

Scientific name	Common name	Death Valley	Mojave % of plots	Lake Mead	Cal-IPC ^a	NatureServe ^a Priority	Reference ^b
Annual grass							
<i>Bromus rubens</i>	red brome	44	80	49	High	–	Abella et al. (2012b)
<i>Schismus</i> spp.	Mediterranean grass	9.1	29.8	45	Limited	High	Cox and Allen (2008)
<i>Bromus tectorum</i>	cheatgrass	15	19	2.6	High	High	Brisbin et al. (2013)
<i>Polypogon monspeliensis</i>	annual rabbitsfoot grass	0.6	1.3	1.2	Limited	Unknown	Halvorson and Guertin (2003)
<i>Bromus berteroi</i>	Chilean chess	0.3	0.0	0.8	–	–	Kemp and Brooks (1998)
<i>Hordeum murinum</i>	mouse barley	0.5	0.2	0.4	Moderate	Unknown	Davison (1971)
<i>Vulpia bromoides</i>	brome fescue	0.0	0.0	0.2	Watch	–	Leys et al. (1991)
Annual-perennial grass							
<i>Bromus diandrus</i>	ripgut brome	0.2	1.3	0.2	Moderate	–	Allen et al. (2005)
Perennial grass							
<i>Cynodon dactylon</i>	Bermuda grass	0.8	0.5	1.4	Moderate	Medium	Mau-Crimmins (2007)
Annual forb							
<i>Brassica tournefortii</i>	Sahara mustard	0.0	3.0	9.9	High	Unknown	Abella et al. (2013)
<i>Salsola tragus</i>	prickly Russian thistle	5.9	2.8	2.6	Limited	–	Beckie and Francis (2009)
<i>Malcolmia africana</i>	African mustard	0.8	0.8	4.3	–	–	Abella et al. (2009)
<i>Sisymbrium irio</i>	London rocket	0.0	0.5	1.4	Moderate	Unknown	DiTomaso et al. (2013)
<i>Salsola paulsenii</i>	barbwire Russian thistle	0.5	0.2	0.0	Limited	Low	Beatley (1973)
<i>Sonchus asper</i>	spiny sowthistle	0.5	0.0	0.2	Watch	–	Hutchinson et al. (1984)
<i>Sonchus oleraceus</i>	common sowthistle	0.0	0.0	0.6	–	–	Widderick et al. (2004)
<i>Portulaca oleracea</i>	little hogweed	0.0	0.2	0.0	–	–	Miyanishi and Cavers (1980)
<i>Xanthium strumarium</i>	rough cocklebur	0.2	0.0	0.0	–	–	Weaver and Lechowicz (1998)
Annual-biennial forb							
<i>Erodium cicutarium</i>	redstem filaree	9.6	62	44	Limited	Medium	Steers and Allen (2010)
<i>Descurainia sophia</i>	herb sophia	0.5	0.5	0.0	Limited	Medium	Best (1977)
<i>Sisymbrium albidissimum</i>	tall tumblemustard	0.3	0.0	0.4	–	–	Howard (2003)
Annual-perennial forb							
<i>Melilotus officinalis</i>	sweetclover	0.5	0.0	0.2	–	Medium	Turkington et al. (1978)

Scientific name	Common name	Death Valley	Mojave % of plots	Lake Mead	Cal-IPC ^a	NatureServe ^a Priority	Reference ^b
Annual grass							
<i>Medicago sativa</i>	alfalfa	0.3	0.0	0.0	–	Insignificant	Sullivan (1992)
<i>Malva neglecta</i>	common mallow	0.0	0.0	0.2	–	Unknown	Elkoca et al. (2005)
Perennial forb							
<i>Marrubium vulgare</i>	horehound	0.2	0.0	0.0	Limited	Medium	Weiss (1996)
Shrub							
<i>Nerium oleander</i>	oleander	0.2	0.0	0.2	Watch	Low	Herrera (1991)
Tree							
<i>Tamarix ramosissima</i>	saltcedar	1.4	2.0	7.3	High	High	Shafroth et al. (2005)
<i>Tamarix aphylla</i>	Athel tamarisk	0.3	0.5	0.8	Limited	–	Walker et al. (2006)

^aSpecies prioritization by the California Invasive Plant Council (Warner et al. 2003, with rankings available from <http://www.cal-ipc.org/>) and NatureServe (Randall et al. 2008, with rankings available from NatureServe Explorer, <http://www.natureserve.org/explorer>). In the Cal-IPC system, species ranked as ‘high’ have greatest potential for invasiveness and negative ecological impacts, ‘moderate’ has intermediate potential, ‘limited’ has least potential, and ‘watch’ represents species requiring additional observation. In the NatureServe system, species are ranked as ‘high’, ‘medium’, ‘low’, or ‘insignificant’, in decreasing order of potential for negatively impacting indigenous biodiversity. ‘Unknown’ represents species evaluated but uncertain. For both ranking systems, ‘—’ signifies that species has not been ranked.

^bReference for either a species’ autecology, regional distribution, or response to management intervention, depending on available articles.

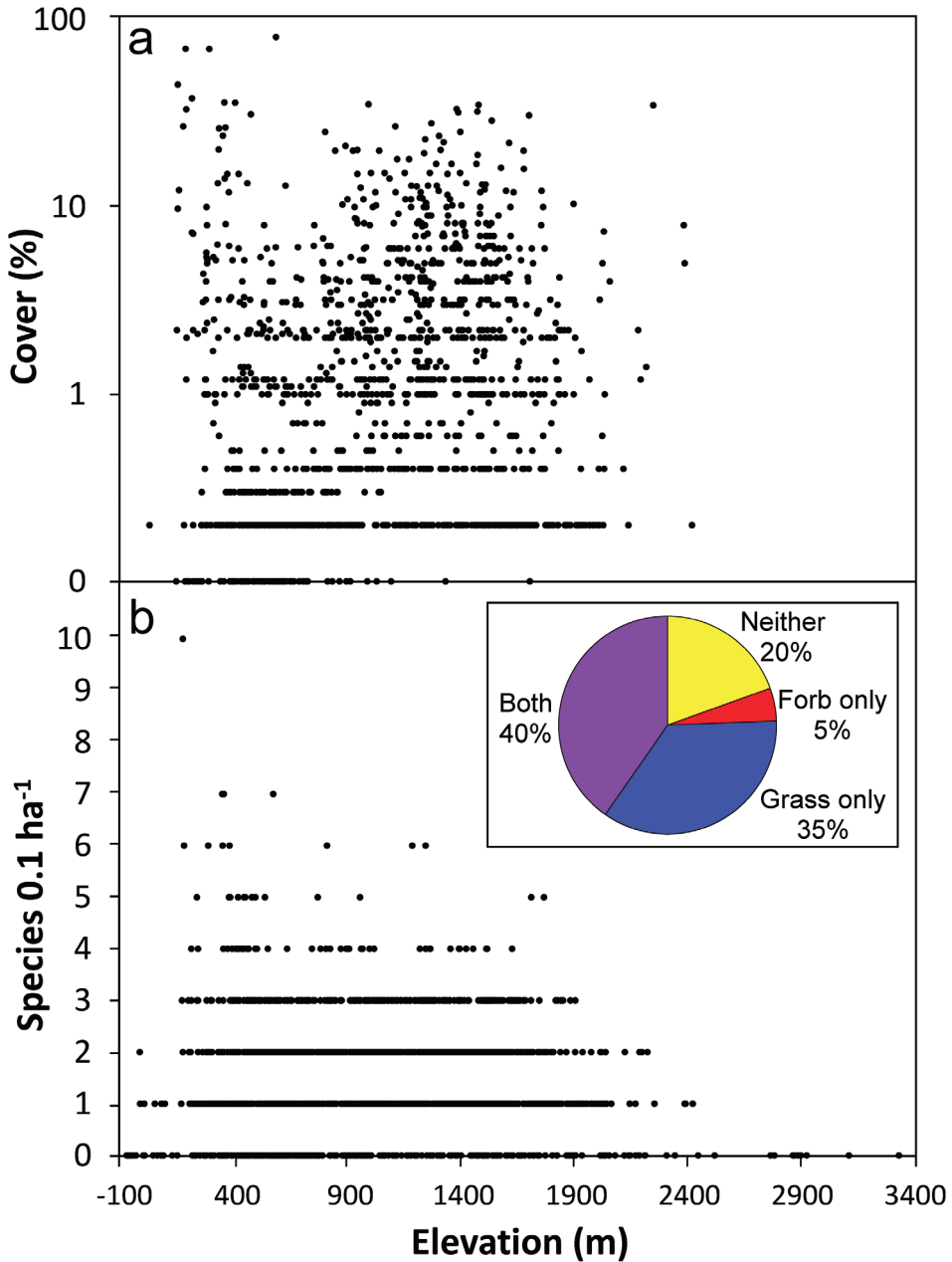


Figure 4. Scatterplot of elevation and non-native plant species richness and cover derived from 1,662 plots in Death Valley National Park, Mojave National Preserve, and Lake Mead National Recreation Area in the Mojave Desert, southwestern USA. There was no relationship between elevation and non-native richness or cover (Pearson $r = 0.00$). The inset graph in (b) shows percentages of plots infested by either non-native grasses or forbs, or both ('neither' signifies 18% of plots not invaded and 2% invaded only by woody species).

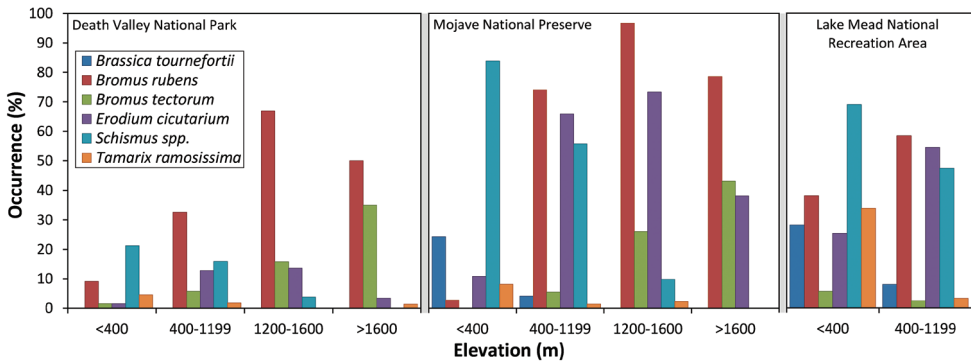


Figure 5. Relationship between elevation and frequency of major non-native plant species in three Mojave Desert parks, southwestern USA. High elevations were absent in Lake Mead.

Table 3. Sørensen similarities of non-native species composition among parks for plots (no parentheses) and park species lists (inside parentheses) in the Mojave Desert, USA.

	Death Valley	Mojave
	Similarity (%)	
Mojave	73 (51)	–
Lake Mead	57 (51)	76 (60)

twice as high in Mojave National Preserve and Lake Mead National Recreation Area as in Death Valley National Park. Mojave Preserve had the fewest un-invaded plots and 3× as many plots containing at least two non-native species as did Death Valley.

Bromus rubens was the most frequent species in all three parks, and its highest frequency was in Mojave National Preserve (Table 2). Notable differences in species frequencies included low frequency of *Schismus* spp. and *Erodium cicutarium* in Death Valley relative to the other parks, and higher frequency of *Bromus tectorum* in Death Valley and Mojave compared to Lake Mead. Additionally, *Brassica tournefortii* was not detected on any plots in Death Valley, whereas the species was the fourth most frequent at Lake Mead.

Park species lists

Lists maintained by each park contained similar numbers of non-native species, ranging from 73–83 species (Table 1). The percentage of a park’s non-native flora detected on plots was also similar among parks at 23–30%. Most species on plots were on these lists except for some new records that the plots produced: Chilean chess (*Bromus berterianus*) and alfalfa (*Medicago sativa*) in Death Valley, and common mallow (*Malva neglecta*) in Mojave Preserve. Compositional similarity of non-native species lists was 51–60% among parks, slightly lower than the 57–76% similarity for plots among parks (Table 3).

Discussion

This assessment suggested that: (i) the parks contain relatively few frequent species, yet these frequent species, mostly annuals, are present on most of the landscape; (ii) non-native plant composition was similar among parks, but non-native frequency was greatest in Mojave National Preserve; (iii) existing species prioritization systems ranked 80% of species and were not always consistent; (iv) over half (51%) of plots contained multiple non-native species; and (v) only elevation extremes tended not to harbor multiple non-native species.

Invasion status

Present invasion status of these parks could be interpreted from different viewpoints. On one hand, the fact that 82% of plots were invaded by at least one non-native species is alarming. Moreover, the ecosystem engineer, *Bromus rubens*, occurred in 60% of plots. By providing copious and persistent fuel, this species promotes spread of wildfire, a novel disturbance requiring centuries for recovery of mature perennial communities in this desert (Abella 2010, Steers and Allen 2010). Some other frequent species, such as *Tamarix ramosissima*, also can dramatically impact indigenous ecosystems, including riparian areas which are hotspots of native biodiversity (Shafroth et al. 2005). On the other hand, 60% of plots had < 1% cover, indicating extensive minimally infested area.

Plot-based surveys of landscapes like ours provide information on species distribution and abundance and are not exhaustive botanical inventories (Barnett et al. 2007). Our plots contained 21% of the 139 non-native species on inventory lists of these parks. Our study years were near average for detecting annual plants, based on receiving 132% of long-term average (74 years) precipitation for the 2010 growing season (October 2009 through April 2010) and 104% for the 2011 growing season (October 2010 through April 2011; Las Vegas, Nevada airport station). Many of the undetected species are uncommon in the backcountry and inhabit only specific sites, such as campgrounds, roadsides, and cultural areas (e.g., historical cabins including non-native landscaping vegetation, or orchards).

Although non-native species measures such as total species and species composition were generally similar among parks, some notable differences existed. Mojave National Preserve had the fewest un-invaded plots, and Death Valley National Park had the lowest non-native richness/plot and fewest plots containing multiple species. Mojave Preserve has the most extensive history of disturbance and was most recently placed under National Park Service protection in 1994 (Beever and Pyke 2005). In the Czech Republic, the later reserves were created, the more non-native plants they contained (Pyšek et al. 2003). Mojave Preserve also has extensive middle elevations most susceptible to non-native plant fuel production and wildfire spread (Van Linn et al. 2013). Less invasion in Death Valley might relate to the park containing elevation extremes, which were least invaded, and the lowest frequencies of *Schismus* spp.

and *Erodium cicutarium*. Another difference was that Lake Mead contained the lowest frequency of *Bromus tectorum*, likely because high elevations were absent (Abella and Tendick 2013).

Elevation

Although correlations between elevation and non-native richness and cover were not detected, individual species were most frequent within particular elevation ranges. Additionally, elevation extremes (below sea level and > 2,000 m) were least invaded in terms of non-native species richness. If climate becomes warmer and drier in the region as some projections suggest (Barrows and Murphy-Mariscal 2012), lower elevations might become even less invasible, and higher elevations more so. Forecasting how invasibility might change at high elevations is difficult, because high elevations may already be invasible and simply have not received seed pressure (Keeley et al. 2003). In Rocky Mountain National Park, Colorado, *Bromus tectorum* frequency increased by 50% across high-elevation ecotonal plots over a 12-year period from 1996–2007 that was relatively dry (Bromberg et al. 2011). Given minimal invasion at the highest elevations in our study parks, early detection and treatment of new invaders and newly invaded sites might be particularly appropriate.

Non-native plant impacts: the desert tortoise and species evolution

Unfortunately, the most frequently detected species, such as *Bromus rubens*, are not simply ‘innocuous’ inhabitants of the parks, but rather the most damaging type of non-native species (i.e. ecosystem engineers; Crooks 2002). These non-natives can disrupt critical ecological functions, fundamentally conflicting with the national park goal of promoting native species and processes. One such example warranting further attention is food availability to the desert tortoise (*Gopherus agassizii*). This long-lived (~ 50 years) reptile inhabits all three parks and is listed as threatened under the U.S. Endangered Species Act. Two studies in the Mojave Desert reported that despite being among the most abundant plants, the non-native *Schismus* spp. and *Bromus rubens* were avoided by foraging tortoises (Jennings 1997, Oftedal et al. 2002). Although *Schismus* spp. represented 98% of the plants encountered, tortoises ate < 0.1% of them (Oftedal et al. 2002). Compared to native annual forbs, *Schismus* has lower water and protein content and high potassium toxicity. Moreover, the pointed florets of *Bromus* can injure tortoises directly when ingested (Medica and Eckert 2007). Non-native annuals compete with native annual forage plants, and natives have increased when *Schismus* and *Bromus* were removed (Brooks 2000). Declining populations of desert tortoises face pervasive dominance of non-preferred food plants, which further influence habitat conditions by providing fuel facilitating spread of desert wildfires (Brooks and Berry 2006).

Our results showing how widespread invading species are on these landscapes exemplify a broad issue of biological invasions being a driver of contemporary species evolution (Leger and Espeland 2010). Two examples from our set of invading species illustrate this. Within 7 years of being exposed to elevated CO₂, *Bromus rubens* plants evolved lower rates of leaf stomatal conductance, a physiological adaptation linked with improved water-use efficiency (Grossman and Rice 2014). Most of the invading species in southwestern deserts are annuals, such as *Bromus*, able to rapidly evolve (Table 2). This underscores the importance of management actions to reduce population sizes of non-natives and their capacity for evolving traits that make them even more competitive (Leger and Espeland 2010). Genetics of native species may also be shifting in response to environments altered by non-native plants. In the Great Basin Desert, growth of native perennial grasses responded most rapidly to watering on sites that were most heavily invaded by *Bromus tectorum* (Leger 2008). This implied that native plants were adapting to become more competitive with *Bromus* (Leger 2008). Adaptation to the presence of a non-native species may be beneficial for persistence of some native plants. However, it is undesirable from a national park perspective, where native species are supposed to evolve through natural processes, not through anthropogenic species introductions.

Implications for species mapping and management planning

These assessment data are an initial step towards non-native plant distribution mapping, which needs to consider extreme spatio-temporal variability in desert ephemeral plants. Distribution and abundance of annual plants varies both with inherent site productivity and weather in any particular year (Wallace and Thomas 2008, Casady et al. 2013). In addition to being a practical strategy given difficulty in sampling numerous desert sites in a short spring growing season, we included live and dead annual plants to both represent cumulative recent ‘presence’ and importance of live and dead biomass as fuel. Thus, the survey data could facilitate spatial modeling of site productivity for these species across the landscape, and serve as baseline data for modeling temporal variation.

Our findings revealed several considerations regarding species prioritization as a management tool. Not all species of management interest had ‘off the shelf’ rankings available, necessitating that managers develop their own rankings, a difficult task for little-studied species. Even if a species has no ranking available, existing ranking systems may still offer a useful framework for developing customized rankings. Results also suggested that comparing different ranking systems, when available, is useful to assess consistency of rankings (Andreu and Vilà 2010). Rankings can differ for numerous reasons, such as different emphases (e.g., inclusion or exclusion of management difficulty as an evaluation factor), and date of the ranking which affects information available. As one example, our comparisons illustrated that *Schismus* spp. were ranked oppositely (‘low’ and ‘high’ priority) by two ranking systems (Randall et al. 2008,

Warner et al. 2003). In addition to providing fuel for wildfire and competing with native plants (Abella and Smith 2013), the observation that *Schismus* is non-preferred forage for the desert tortoise suggests that the 'low' ranking for *Schismus* warrants re-evaluation (Ofstedal et al. 2002).

Prioritizing species currently at the extremes – those that are infrequent (but capable of impacts) and those that are widespread and capable of major impacts – may maximize use of limited treatment resources. For example, early treatment of currently infrequent species such as riggut brome (*Bromus diandrus*) and spiny sowthistle (*Sonchus asper*) would follow a principle that early detection and treatment is the most cost-effective and successful strategy (Lodge et al. 2006). This also helps reduce risk that these species become future problems.

Rather than viewing pervasive, high-impact invaders like *Bromus rubens* as 'hopeless', treating these species at priority sites is likely important to avoid negating other management efforts and indeed protecting core values of parks. Over 28,700 ha (4.5%) of Mojave National Preserve burned in fires partly fueled by *Bromus rubens* between 2005 and 2011, destroying mature desert vegetation, as well as cultural resources (Hegeman et al. 2014). Strategically treating priority sites can reduce landscape fuel connectivity (Brisbin et al. 2013). Treatments suitable for broad areas such as those infested by *Bromus rubens* require further experimentation, but early timed herbicide application has reduced *Bromus* while promoting natives (Allen et al. 2005). Competitive native species also can reduce *Bromus* (Abella et al. 2012), and biocontrol agents are under evaluation (Baughman and Meyer 2013).

We identified sites containing multiple non-native plant species, which can affect candidate treatment strategies and their effectiveness. The potential influences of multiple species are numerous, such as: (i) herbicide effectiveness can vary with plant growth form, (ii) treatment timing can be difficult when species' phenologies differ, (iii) required treatment duration can fluctuate among species varying in soil seed bank longevity, (iv) more complicated treatment regimes can increase costs and potential for negatively impacting native species, and (v) chances increase that other non-native species replace a focal treated species (Abella 2014).

What evidence exists for relationships of multiple species with treatment difficulty in the Mojave Desert? Brooks (2000) found that hand pulling *Bromus rubens* or *Schismus* spp. increased native annuals but also increased the non-native forb *Erodium cicutarium*. Similarly, hand weeding *Brassica tournefortii* increased *Erodium* (Marushia et al. 2010). In a post-fire environment dominated by non-native annuals, Steers and Allen (2010) had more encouraging results where herbicide not only reduced the grasses *Bromus* and *Schismus*, it also reduced *Erodium*. Native annual forbs increased, and the native grass sixweeks fescue (*Vulpia octoflora*) was not damaged (Steers and Allen 2010). The treatments exploited the accelerated early season phenology of non-native annuals (compared to native annuals) by applying herbicide early in the growing season. Refining knowledge of the earliest possible time for treating non-natives – which might vary among years – is warranted to manage single- and multi-species infestations while promoting natives (Marushia et al. 2010, Abella et al. 2013).

Conclusion

Invasion by non-native species is generally inconsistent with national park objective of conserving native species and ecological processes (National Park Service 2006). Our assessment illustrated that non-native plants infested 82% of 1,662 plots in three of the four largest national parks in the contiguous USA. Given numerous priorities for management, how much attention do non-native plants warrant? We suggest that greater attention is warranted, because non-native plant invasions can impact essentially all functions of parks, ranging from nature conservation to visitor experiences and viewsheds (Lodge et al. 2006). Furthermore, non-native species can interact with other stressors such as climate change. In Joshua Tree National Park in the southern Mojave Desert, for example, Barrows and Murphy-Mariscal (2012) reported that projected future habitat of the park's namesake (Joshua tree, *Yucca brevifolia*, a fire-susceptible species) in a changing climate would continue to correspond with that of the fire-promoting, non-native annual *Bromus rubens*. With continuation of novel fire regimes, it is unclear how many *Yucca* trees may still be around to even experience a future climate.

At least three strategies may facilitate reducing non-native plant invasion. First, given the limited and short duration of funding allocated to treating non-native plants, 'institutionalizing' non-native plant management in park operations is likely critical. For example, infusing knowledge of non-native plants into visitor education and developing systems for park staff and visitors to report infestations while moving through parks can be cost-effective (Crall et al. 2012). Second, owing to interrelatedness of non-native species and other stressors, treating non-natives while managing other stressors (e.g., nitrogen deposition via air pollution) may increase ecological effectiveness and cost-efficiency. Third, dedicating further resources to managing non-native plants may be essential to protect core park values. It should be recognized that dedicating resources to treating non-native plants in most national parks is recent, such as formation in 2000 of the National Park Service's Exotic Plant Management Teams. Moreover, enormous parks such as the ones of our study do not necessarily receive more resources than parks < 1% their size, and this size:resource imbalance may require attention. Existing non-native plant invasion is unlikely to represent a 'hopeless' situation for native species conservation in our study parks, because 60% of plots had < 1% cover of non-native plants.

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