SUBMISSION TO THE EPBC ACT REVIEW

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Organisation
The Nature Conservancy

Attachment provided?
Yes

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SUBMISSION RESPONSES

This submission was provided as an attachment only. The attachment is provided on the following pages of this document.
17 April 2020

EPBC Act Review Secretariat  
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Dear Professor Samuel

The Nature Conservancy welcomes the opportunity to provide input into the 2020 review of Australia’s Environment Protection and Biodiversity Conservation Act 1999.

The Nature Conservancy is one of the world’s largest conservation organisations, working around the world to conserve the lands and waters on which all life depends. Founded in 1951, we work in 70 countries across 6 continents. Since establishing a program in Australia in 2002, The Nature Conservancy has collaborated with a wide array of partners to support some of the most pressing conservation issues across the country. This includes contributing to achieving many of our international obligations, from restoring endangered ecosystems, building the protected area estate and using natural solutions to tackle climate change.

The Nature Conservancy has been an active participant in developing policy around these issues, providing funding and developing innovative finance mechanisms for conservation and building capacity for Indigenous, coastal and farming communities for managing conservation values in their landscapes. Strong, well-implemented environmental legislation is a critical underpinning of safeguarding biodiversity and confidence for both public private investment in biodiversity conservation.

We welcome the opportunity to comment on the ten-year review of the Environment Protection and Biodiversity Conservation Act 1999 and respond to the questions posed in the discussion paper below.

We attach a number of papers providing more details on some of the models proposed and The Nature Conservancy’s work in Australia that is relevant to reviewing the EPBC Act. We would be happy to discuss and provide further information on any of the points below.

Yours sincerely

Dr James Fitzsimons  
Director of Conservation and Science
Response to questions posed in the EPBC Act Review discussion paper

Question 1 - Some have argued that past changes to the EPBC Act to add new matters of national environmental significance did not go far enough. Others have argued it has extended the regulatory reach of the Commonwealth too far. What do you think?
If the objectives of the Act are to be achieved, the matters of national environmental significance need to be expanded to encompass more of Australia’s environment. This is outlined in more detail in responses to Questions 2, 3 and 4 below.

Question 2 - How could the principle of ecologically sustainable development (ESD) be better reflected in the EPBC Act? For example, could the consideration of environmental, social and economic factors, which are core components of ESD, be achieved through greater inclusion of cost benefit analysis in decision making?
If the overall aim of the Act is to ‘protect and conserve Australia’s environment, biodiversity and heritage and promote ESD through the conservation and sustainable use of natural resources’, its current list of provisions do not enable it to do so for large parts of the country. This is in part due to a limited range of matters of national environmental significance (e.g. threatened species and ecological communities and opposed to all species and all ecological communities)\(^1\). This means the Act usually only is triggered when dealing with species or systems in decline as opposed to more proactive use of the Act in landscapes that are still intact but face development pressure. This is a key area for reform if the Act is to achieve its aims (see also Question 4 below).

Certainly, benefit:cost analysis could be included in decision making as part of a broader suite of decision-support tools. However, until investment in environmental economic accounting and quantifying ecosystem services is increased, the environment will typically be at a disadvantage in a benefit:cost analysis because it will suffer from poorer data sources.

More broadly the Act should support processes of decision-making that promote full understanding of the environmental, social and economic outcomes of development decisions, and enable clear articulation of the implications of each decision (and the cumulative impact of decisions) and trade-offs. More attention should be given to understanding community aspirations and needs in this process, with principles of equity and understanding public and private interests at the core of decision-making processes. Specific attention should be paid to the aspirations of Indigenous and communities and the cultural and Indigenous knowledge aspects of ESD.

Question 3 - Should the objects of the EPBC Act be more specific?
No, these are objects that one would expect in a broad piece of environmental legislation. However, provisions should reflect these objects and not focus on a narrower subset as they currently do (e.g. observation on MNES above).

Question 4 - Should the matters of national environmental significance within the EPBC Act be changed? How?
Current matters of national environmental significance are somewhat ad hoc, reflecting some clear obligations under international treaties (e.g. world heritage areas, Ramsar sites), some elements of other treaties (e.g. threatened species and communities but not other elements of the Convention

\(^1\) Importantly, the objects of the Act refer to “especially those aspects of the environment that are matters of national environment significance” but this should not be interpreted as ‘exclusively’ or ‘solely’.
on Biological Diversity) carried over from previous legislation with some eclectic additions of single places (GBRMP) or threatening processes (water trigger) since 1999.

This mix means some species or parts of the landscape receive different levels of protection based on this historical legacy and not necessarily on conservation value or relative threat. For example, northern Australia contains the largest intact tropical savanna on earth, central Australia the largest intact desert landscape on earth, and the Great Western Woodlands of south-west Australia the largest intact temperate woodland, yet these values are not recognised under MNES and indeed due to their intactness receive little protection under the EPBC Act².

The Act should be amended to include:
- ecosystems with a high importance for delivering ecosystem services (such as carbon and water)
- rare or regionally important ecosystems
- recognition of areas and processes that support healthy, resilient ecosystems and connectivity, areas of relatively high intactness, and ecosystems that will support natural climate solutions
- environmental assets of significance to indigenous peoples, both to recognise and support international obligations relating to the rights of indigenous peoples and to take into account important indigenous knowledge and capacity for protection and management

Question 5 - Which elements of the EPBC Act should be priorities for reform? For example, should future reforms focus on assessment and approval processes or on biodiversity conservation? Should the Act have proactive mechanisms to enable landholders to protect matters of national environmental significance and biodiversity, removing the need for regulation in the right circumstances?

In line with expanding definitions of MNES above, greater application of the Strategic Assessment process would be beneficial. Strategic Assessments are an important mechanism to plan development and biodiversity conservation at a landscape scale for the long term, reducing cumulative, ‘death-by-a-thousand-cuts’ development and enabling proactive conservation management and planning at a scale that is relevant to species and ecological communities. Greater integration of bioregional planning principles into strategic assessments would be welcome. Increased incorporation of Indigenous cultural values into such integrated assessments is also important, particularly in landscapes where Indigenous ownership and/or Native Title rights exist over land. For example, The Nature Conservancy in partnership with the Nyikina Mangala Traditional Owners in the Kimberley, undertook Development by Design planning which incorporated ecological and social values identified the community for their country as part of a Healthy Country Planning process, then spatially mapped these values in order to allow the community to make informed decisions on development potential³. The use of Healthy Country Planning has been popular in many Indigenous communities across Australia⁴ and there is growing demand for Development by Design

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planning. This process could be coupled with a more proactive strategic assessment program to improve Indigenous cultural outcomes.

In relation to the question above, while more proactive mechanisms (such as incentive funding and MNES matters protected through conservation agreements or other state-based conservation covenants) would be valuable to increase protection, these should not be at expense of regulation. It is well accepted that both incentives and regulation are needed to offer the best outcomes as it will cover both the entire landholder base (through regulation) and reward those undertaking positive actions (incentives). The community-based conservation and development planning tools mentioned above can play an important role in supporting both regulatory and incentive mechanisms, as well as contributing to informed decision-making and Free, Prior and Informed Consent requirements.

Australia has a relatively high take up of private land conservation mechanisms from permanent, statutory conservation covenants\(^5\), set term conservation agreements\(^6\) and non-binding programs such as Land for Wildlife and Landcare\(^7\). However, at present, there are multiple inhibitors in Australia’s tax system that inhibit proactive biodiversity conservation on private land\(^8\) and these need to be considered in reviewing the EPBC Act and conservation programs.

**Question 6 - What high level concerns should the review focus on?** For example, should there be greater focus on better guidance on the EPBC Act, including clear environmental standards? How effective has the EPBC Act been in achieving its statutory objectives to protect the environment and promote ecologically sustainable development and biodiversity conservation? What have been the economic costs associated with the operation and administration of the EPBC Act?

Considering broad-scale land clearing, a legally preventable threatening process has still been occurring in Australia over the past decade, it is hard to justify an assessment that the Act is achieving its statutory objectives to protect the environment and promote ecologically sustainable development and biodiversity conservation. As broad-scale land clearing is one of the greatest causes of biodiversity loss, the Act needs to explicitly address this. While there may be current limitations based on how MNES are determined, having provisions to address broad-scale land clearing should be one of the highest priority areas for reform.

Streamlining and rapid assessments of potential threatened species and ecological communities should be a priority. It is commonly accepted that the number of species and ecosystems that are actually threatened is greater than those listed under the EPBC Act. As such, these unlisted species and ecosystems are not receiving the protection they are entitled to. Peer-reviewed assessments using the IUCN Red List for either species or ecosystems could enable rapid addition to the schedules in the Act. The IUCN methodology is already accepted as part of the *Intergovernmental Memorandum of Understanding Agreement on a Common Assessment Method for Listing of Threatened Species and Threatened Ecological Communities*. For example, a recent peer-reviewed published assessment of the conservation status of the Oyster Reef Ecosystem of Southern and

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Eastern Australia shows this system is critically endangered\(^9\), and this should enable fast track listing, providing protection to this ecosystem and increased prioritisation for recovery funding.

Another area for reform is to support genuinely informed decision-making that considers the full range of environmental, cultural and socio-economic values impacted by development decisions, and enables local people, communities and wider interests to have appropriate input. Particular attention should be given to indigenous communities in northern and central Australia where much of the land is held under various forms of Indigenous title and where community aspirations need to be much better considered and supported.

The Act should apply to forests covered in Regional Forest Agreements (RFAs). The continued decline of a number of threatened species in forested landscapes highlights the need to ensure the Act applies equally to all landscapes/ecosystem types. RFAs (and the Comprehensive Regional Assessments originally used to inform them) are now relatively old and despite a renewal process in many states, these have not had the same level of public input nor science as the original process. Our knowledge of species ecology, their habitat and location, and threats has improved since Comprehensive Regional Assessments were developed over 20 years ago.

**Question 7 - What additional future trends or supporting evidence should be drawn on to inform the review?**

The impacts of climate change will have important consequences for the assets the Act is trying to protect and the way the Act needs to respond to impacts and events associated with climate change. Longer and more frequent droughts and more frequent and intense bushfires are two areas likely to place increased pressure on threatened species and ecological communities and on places such as Ramsar and World Heritage sites. The concept of defined ecological communities and their condition may change as climate change potentially alters the composition of these ecosystems over time. Acknowledging this change will be important to ensure this is incorporated into recovery plans and for considerations of future habitat potential and species/communities ranges over the decades to come.

Natural systems are one of the best storers of carbon. This needs to be recognised in the Act and provisions made to ensure this function is not lost through land use change. In addition, there is growing interest in using and developing nature-based solutions for problems that might have otherwise relied on typical grey infrastructure\(^10\). Better incorporation of nature-based solutions in approvals for developments should be considered as part of the review of the Act.

Intensification of land use is likely in many systems\(^11\). Without active consideration and acknowledgement of this trend, some systems will be increasingly inhospitable for a range of species and ecosystems, even if such activities do not trigger MNES as currently defined. While mechanisms such as strategic plans could be one way of more systematically considering and regulating land use change/intensification, these mechanisms need to apply more widely than what they have been to date.

The Act should also recognise the increasing role and significance of indigenous land managers across large areas of northern and outback Australia and their importance in protecting, restoring

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\(^10\) [https://www.nature.org/content/dam/tnc/nature/en/documents/NBSWhitePaper.pdf](https://www.nature.org/content/dam/tnc/nature/en/documents/NBSWhitePaper.pdf)

and maintaining biodiversity values on behalf of all Australians. Indigenous knowledge and capacities for long-term monitoring and reporting across remote large landscapes should be supported.

**Question 8 - Should the EPBC Act regulate environmental and heritage outcomes instead of managing prescriptive processes?**

The discussion paper suggests “This regulation is resulting in unnecessary uncertainty and delays with flow on impacts to industry, governments and the community” with an implication that it is an inherent problem with the Act and associated Regulations. While most would agree, improved efficiency and timeliness of decisions (where possible) is important, adequate resourcing of the responsible Department to process and make the assessments in a timely manner is critical if the balance between speed of decision making and protection of the environment is to be achieved.

In relation to the question posed, the Commonwealth should have a role in both the setting of broader outcomes and standards as well as a process of assessment and decision-making. See also response to Question 14.

**Question 9 - Should the EPBC Act position the Commonwealth to take a stronger role in delivering environmental and heritage outcomes in our federated system? Who should articulate outcomes? Who should provide oversight of the outcomes? How do we know if outcomes are being achieved?**

The Commonwealth has in many cases been successful when taking a strong role in delivering environmental and heritage outcomes. A good example of this was the National Reserve System Program that saw the Federal Government drive an increase in the growth of the protected area estate, with a particular emphasis on increasing the representation of under-represented bioregions and ecosystems. This was achieved through a dedicated fund, smart leveraging of non-Commonwealth Government sources (the Commonwealth paying up to two-thirds of the purchase price, and the state or NGO the remaining one-third) and a cooperative, science-based approach, underpinned by good, nationally-accepted policy which contributed directly to Australia’s international obligations under the Convention on Biological Diversity. While the EPBC Act was not a driver of this work it could have been used to ensure Conservation Agreements (an under-utilised provision under the Act) were signed over purchased land that did not go into the public protected area estate, but were held, mostly by land trusts, as privately protected areas. Alternatively, land purchased through this program could have become a new MNES (much like Ramsar sites and World Heritage Areas) to reflect the national importance of these new protected areas and the Commonwealth’s role in establishing them. Both protection options could have prevented some of the development threats (or at least given the Commonwealth a greater role in the approval process) that were subsequently realised on a small number of these properties.

Expanding the delivery of environmental outcomes (such as through the National Reserve System Program) is an important opportunity that fulfils both the objectives of the Act and our international obligations under the Convention on Biological Diversity. However, the key to the success of the program was a dedicated fund at a scale to match the challenge and a willingness to innovate. Simply having the EPBC Act give the Commonwealth Government an obligation to create a comprehensive, adequate and representative reserve system will not make it happen in the absence of these other elements.

The above example highlights that good policy and programs with dedicated funding can achieve significant national results and could also benefit with extra provisions and protection under the EPBC Act.
**Question 10** - Should there be a greater role for national environmental standards in achieving the outcomes the EPBC Act seeks to achieve? In our federated system should they be prescribed through: Non-binding policy and strategies? Expansion of targeted standards, similar to the approach to site contamination under the National Environment Protection Council, or water quality in the Great Barrier Reef catchments? The development of broad environmental standards with the Commonwealth taking a monitoring and assurance role? Does the information exist to do this?

There are a number of ways existing mechanisms under the Act could be improved with clearer, more transparent standards to have an immediate positive impact on biodiversity, as outlined below.

**Recovery Plans and Critical Habitat**

Ensuring clearer standards, outcomes and timely development of recovery plans and the establishment and operation of recovery teams for threatened species and ecological communities is needed to ensure these species and ecological communities have the best opportunities to recover. Many species/ecological communities do not have a recovery plan at present and a logical standard would be to ensure one is in place within a set time from listing (e.g. one year). Secondly, the identification of ‘critical habitat’ within such plans and the relationship to the Register of Critical Habitat requires both greater transparency and standards. At present there are only 5 sites listed under the EPBC Act’s Register of Critical Habitat. The under-utilisation of critical habitat provisions is in stark contrast with the US *Endangered Species Act of 1973* where critical habitat listings have been used extensively and have a marked improvement in the likely recovery of threatened species populations. One approach might be that critical habitat identified by recovery teams or in recovery plans is assessed by the Threatened Species Scientific Committee with subsequent listing on the Register. There is a broader need to review the process for rapidly identifying and listing critical habitat in response to large-scale disturbance events that might make remaining undisturbed habitat highly important in a short space of time. The unprecedented 2019-2020 bushfires of southern and eastern Australia highlights this issue. The size of the area burnt, and intensity of the fire has meant the majority of habitat for a large number of threatened species has been impacted, placing high conservation value on the unburnt refuges remaining. Considering the importance of unburnt refuges, applying the precautionary principle may mean that many or most unburnt areas within or adjoining fire scares would qualify as critical habitat, even without full surveys of these refuges to determine species occupancy or immediate suitability. A more structured approach to critical habitat listing, particularly following situations as experienced in these bushfires (large amounts of habitat burnt for a large number of threatened species and the need for quick action, with, at times, limited data) would bring increased certainty to species recovery efforts, to industry and to private landholders alike, and ensure recovery efforts are best spent where needed. Regardless, reform of the provision relating to critical habitat under the Act is required – specifically, enforceability needs to apply to all land tenures, not just Commonwealth land.

**Biodiversity offsets**

Biodiversity offsets are another area where greater consistency in standards should be applied. While there is overarching guidance for applying offsets under the EPBC Act, the implementation, on-ground outcomes and level of security of those outcomes varies greatly. While strategic

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assessments are one way to achieve more strategic outcomes through offsetting, there is a need to improve the delivery of offsets in a less piecemeal way for areas outside of strategic assessments. At present, it is often left to the proponent to find the offset often resulting in an unstrategic outcome based on what offsetting opportunities are available at the time. Options for ‘pooling’ offsets to achieve a much bigger, more strategic, or more secure outcome than could be achieved by any single offset should be considered. For example, many small offsets comprising revegetation projects scattered through the landscape may have low success, low security and ultimately, little conservation gain, but pooling funds spent by proponents could achieve a larger, secure outcome through, for example, a conservation covenant or land acquisition to add to the reserve estate. This may involve a panel comprising Australian and state government staff, science and other external expertise to assess where such pooled offsets might be realised.

The Nature Conservancy’s experience with community-based planning in support of Indigenous community conservation and development aspirations (using Health Country Planning and Development by Design tools mentioned earlier) shows that better understanding and articulation of values that need to be considered in offset arrangements can be gained through these community-led processes. The Act should better support and recognise informed planning and decision-making processes.

**Question 11 - How can environmental protection and environmental restoration be best achieved together? Should the EPBC Act have a greater focus on restoration? Should the Act include incentives for proactive environmental protection? How will we know if we’re successful? How should Indigenous land management practices be incorporated?**

A combination of good legislation, good policy and good programs are essential to see environmental protection and restoration achieved together. The EPBC Act is a critical piece of legislation for the protection and management of Australia’s natural environment and heritage and plays mainly a regulatory/restrictive role as opposed to proactively requiring restoration to happen. This proactive restoration role has typically been the purview of programs, even if those programs were the result of legislation (e.g. the Natural Heritage Trust of Australia Act 1997). The EPBC Act could incorporate new, proactive measures for restoration, however to be effective, these will need to have or enable funding/financing mechanisms to match the ambition.

The need to incentivise conservation has long been recognised as an important complement to regulation. Incentives on private land have varied from one-off payments for particular management activities or protection designations, mid-term agreements and associated payments for the delivery of particular outcomes (e.g. conservation tenders), to tax deductions and rate relief.

Many incentive programs (e.g. reef protection, water quality protection) have been designed to repair past poor practice. Incentive programs are increasingly recognising values of intact or healthy landscapes that need protection or management for long term health and resilience. The NSW Biodiversity Conservation Trust, a statutory authority set up under the NSW *Biodiversity Conservation Act 2016*, will be investing $240 million over 5 years and ongoing funding of $70 million each following year, with an emphasis on priority ecosystems and species and permanent protection.

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of these assets\textsuperscript{15}. The Craik Review called for a similar fund at a national level\textsuperscript{16}. Funding for long-term management of high priority (be they threatened, under-represented or large intact) ecosystems/species habitat on private and Indigenous land coupled with appropriate protection mechanisms will be critically for achieving the objectives of the Act.

While grants have many advantages and will continue to form a vital piece of the funding mix for conservation and agriculture they are limited by their ‘once and for all’ outlay. This may suit some activities, but environmental bonds have the potential to provide upfront capital and risk-sharing (risk taken on return, with capital guarantee from government). In a proposal developed by The Nature Conservancy in consultation with multiple stakeholders, a model could see the Australian Government work with experts to identify a suite of landscape resilience priorities to be achieved to 2030. To fund the achievement of these outcomes, a special purpose vehicle would issue a 10-year, multi-million dollar bond to Australian superannuation funds and tens of millions in full-risk equity to impact investors. The Bond principal and coupon could be guaranteed by the Commonwealth, with coupon payments commencing in 2025. The impact investors would earn a target rate of return – beginning in 2025 – only if agreed outcomes are delivered.

In addition to the bonds approach above, there is significant potential for a sustainable mechanism to mobilise large-scale private sector investment in nature-based solutions that support a long-term, future focused and resilient farm and forestry sector in Australia. It would catalyse the development of a self-sustaining market through which farmers will be rewarded for stewardship of natural assets, transforming the agriculture sector. We will provide more detail on this proposal separately.

Indigenous land management programs in the relatively intact landscapes of northern and central Australia should be supported as an important component of efforts to maintain intact or restore degraded systems. Indigenous land management practices should be incorporated through:

- stronger support for indigenous-led planning and decision-making process in development impact assessment as described earlier
- recognising information and data gaps that lead to suboptimal decision and support processes to fill these information gaps in assessments and decision-making processes
- greater recognition and support for use of traditional and local knowledge
- specific financial and facilitation support for indigenous involvement in assessment process and similar support for indigenous-led planning and offset negotiations
- support for indigenous involvement in monitoring and evaluation programs (recognising capacity, knowledge and cost-effectiveness especially in remote landscapes)

Greater standards for restoration could be incorporated into various parts of the Act that currently require it (e.g. offsets or restoration efforts for nationally threatened ecological communities). The Society for Ecological Restoration Australasia has developed national standards based on wide consultation\textsuperscript{17} which could usefully be incorporated into national guidelines for the EPBC Act. To improve the conservation status for endangered and critically endangered ecological communities, active restoration will be required.

\textsuperscript{15} https://www.bct.nsw.gov.au/what-we-do
\textsuperscript{17} http://www.seraustralasia.com/pages/standards.html
**Question 13 - Should the EPBC Act require the use of strategic assessments to replace case-by-case assessments? Who should lead or participate in strategic assessments?**

As above, a more systematic roll out of strategic assessments would be welcomed, particularly if they included a broader range of matters of national environmental significance than are currently considered. Much like past assessments, these should be jointly led by Federal and State/Territory governments (the states/territories being primarily responsible for land use planning on land and in coastal waters). Participation should be broader than it has been to date. Greater involvement of Indigenous groups, community groups, farming organisations, NGOs and university scientists (amongst others) should all be considered to get better and more enduring outcomes.

**Question 14 - Should the matters of national significance be refined to remove duplication of responsibilities between different levels of government? Should states be delegated to deliver EPBC Act outcomes subject to national standards?**

The main area of duplication between state and federal governments is through threatened species and ecological community listings. While there is merit in having greater consistency in the listing process to enable this reduction in duplication and/or delegation, this would firstly need involve a commitment to more rapid listing processes, development and implementation of recovery plans, identification of critical habitat and willingness to enforce (most which could be written into the Act). It should also be recognised species may be threatened in some parts of its range (e.g. in some states) but not in others. Thus, recognition of the regional difference would have to be incorporated into the EPBC Act if this duplication was to be removed.

**Question 15 - Should low-risk projects receive automatic approval or be exempt in some way? How could data help support this approach? Should a national environmental database be developed? Should all data from environmental impact assessments be made publicly available?**

Whether low-risk projects receive automatic approval or be exempt would depend on who is determining the level of risk and the level of transparency involved in the decision making. A reliance on automated processes would be contingent on data being comprehensive enough to allow those decisions to be robust. Many areas of Australia are data-poor, so decisions about low-risk projects should be made extremely cautiously, and there would have to be a commitment to filling gaps prior to decisions being made. There are particular risks related to lack of comprehensive assessment of cultural sites and biodiversity values across last areas of Australia. A national environmental database should be developed but a commitment to ensuring this is kept up to date will be important if it is the basis for automated decision making. All data from environmental impact assessments should be made publicly available (except for sensitive species data that may be at greater risk if their location is revealed, and Indigenous cultural data unless communities permit its release). Large amounts of ecological data collected as part of the regulated EIA process simply sits in consultants reports and it should be mandatory that these reports are publicly available, and the data deposited in databases such as the Atlas of Living Australia.

**Question 16 - Should the Commonwealth’s regulatory role under the EPBC Act focus on habitat management at a landscape-scale rather than species-specific protections?**

To achieve the objective of the Act, the focus needs to be on both scales. It is true the strategic assessments have been (and have the increased potential to be in the future) important land use planning instruments, achieving multiple outcomes under the Act. Likewise, the National Reserve System Program (a well-funded and strategic program, but not an instrument of the Act) had been a highly successful proactive instrument in achieving multiple objectives, from international obligations for growing the protected area estate, to protecting threatened species habitat and protecting sites of national and international significance. The National Reserve System Program had
important benefits beyond the objective of the Act by bringing significant new funds to the conservation sector with its leveraged funding approach\textsuperscript{18}. Reinstatement of a dedicated fund, be it as a program or an instrument within the EPBC Act should be a high priority\textsuperscript{19}.

**Question 17** - Should the EPBC Act be amended to enable broader accreditation of state and territory, local and other processes?  
See response to Question 14.

**Question 18** - Are there adequate incentives to give the community confidence in self-regulation?  
See Question 11. Incentives are not currently adequate to consider self-regulation. A comprehensive review of which incentive types have worked for which landholders in which landscapes and the social\textsuperscript{20} and economic\textsuperscript{21} reasons these have been successful (and, just as importantly, where they have not) is needed to ensure incentives are designed to match diverse landholder circumstances and values and thus have increased uptake.

**Question 19** - How should the EPBC Act support the engagement of Indigenous Australians in environment and heritage management? How can we best engage with Indigenous Australians to best understand their needs and potential contributions? What mechanisms should be added to the Act to support the role of Indigenous Australians?  
See response in Question 5. Indigenous people own, have rights to or management responsibilities for some of the most significant natural assets in Australia. Supporting Indigenous people to undertake conservation management is thus a critical part of achieving objectives under the Act and national biodiversity policy. At present, this support is largely in the form of funding for Indigenous ranger positions through Working on Country program, rangers and some management costs through the Indigenous Protected Area (IPA) Program and, particularly in northern Australia, through selling carbon credits earned through savanna burning. Our experience suggests there is demand for new IPAs and new ranger teams and positions, as well as a desire to have longer term continuity of funding for this important work. Like suggestions above in response to Question 11 for private land, a large trust fund would ensure more certain, longer term funding.

**Question 20** - How should community involvement in decision-making under the EPBC Act be improved? For example, should community representation in environmental advisory and decision-making bodies be increased?  
Indigenous community involvement should be strengthened to enable informed decisions-making and full Free, Prior and Informed Consent and ensure that community values and aspirations are properly considered. As described earlier, community-based planning and assessment processes can be used to strengthen these aspects.

Similarly, other forms or local and regional planning can provide important indicators of community values and objectives that need to be considered in assessment process and trade-offs (e.g. local government biodiversity planning or corridor planning, NRM plans etc.). If community representation on decision making bodies relating to the Act, it should be clear their role is to advance the objectives of the Act.

**Question 21 - What is the priority for reform to governance arrangements? The decision-making structures or the transparency of decisions? Should the decision makers under the EPBC Act be supported by different governance arrangements?**

Transparency in decision-making is important and it is important that this is increased. For example, decisions on development proposals should be published as should minutes of Threatened Species Scientific Committee meetings.

**Question 22 - What innovative approaches could the review consider that could efficiently and effectively deliver the intended outcomes of the EPBC Act? What safeguards would be needed?**

Ecosystem services are an important value of natural systems and recognition of their role and importance could have benefits for biodiversity. Although the EPBC Act Review discussion paper suggests ecosystem service markets for carbon is in limited use by the Commonwealth and the states, the use is more extensive than implied with savanna burning and vegetation management making important contributions to carbon abatement and sequestrations under the Emissions Reduction Fund\(^\text{22}\). The take up of savanna burning projects in northern Australia has been one of the largest adoptions of a payment for ecosystem services (in terms of area) in the world, with biodiversity and cultural outcomes\(^\text{23}\). Increased interaction between the EPBC Act and the *Carbon Credits (Carbon Farming Initiative) Act 2011* could see increased vegetation management or ecosystem process methods developed that have a direct positive impact for biodiversity (e.g. a blue carbon methodology which would benefit mangroves, saltmarsh and coastal wetlands). Safeguards for ensuring perverse outcomes (such as a greater emphasis on achieving carbon production over biodiversity in any one location) are avoided, will also be important.

**Question 23 - Should the Commonwealth establish new environmental markets? Should the Commonwealth implement a trust fund for environmental outcomes?**

The Commonwealth could support the development of a biodiversity credit market that would allow for corporate and philanthropic investment to deliver environmental outcomes. This could be done through providing the necessary policy and regulatory enabling environments or through the Commonwealth establishing markets itself. Other markets, such as a nutrient trading market are emerging to reduce damaging nutrient import into places such as the Great Barrier Reef. These could also have positive flow-on benefits for terrestrial land use through habitat retention and restoration. In any market design, careful consideration for perverse outcomes will be important to ensure environmental benefits gained in one area do not negatively impact on others. In addition, utilising existing markets that do not have an explicit environmental focus could also have positive outcomes for conservation. For example, the Murray-Darling Basin Balanced Water


businesses and governments” has not been supported by compelling evidence in the discussion paper to justify including as part of the principle. While making decisions simpler (and timelier) should be an objective, improved resourcing of the administration of the Act could potentially achieve this, rather than removing regulations.
Short communication

Agricultural intensification and loss of matrix habitat over 23 years in the West Wimmera, south-eastern Australia

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Agricultural intensification

ABSTRACT

The global trend toward more intensive forms of agriculture is changing the nature of matrix habitat in agricultural areas. Removal of components of matrix habitat can affect native biota at the paddock and the landscape scale, particularly where intensification occurs over large areas. We identify the loss of paddock trees due to the proliferation of centre pivot irrigation in dryland farming areas as a potentially serious threat to the remnant biota of these areas. We used a region of south-eastern Australia as a case study to quantify land use change from grazing and dryland cropping to centre pivot irrigation over a 23-year period. We also estimated rates of paddock tree loss in 5 representative landscapes within the region over the same period. The total area affected by centre pivots increased from 0 ha in 1980 to nearly 9000 ha by 2005. Pivots were more likely to be established in areas which had originally been plains savannah and woodlands containing buloke (\textit{Allocasuarina luehmannii}), a food source for an endangered bird. On average, 42\% of paddock buloke trees present in 1982 were lost by 2005. In the two landscapes containing several centre pivots, the loss was 54\% and 70\%. This accelerated loss of important components of matrix habitat is likely to result in species declines and local extinctions. We recommend that measures to alleviate the likely negative impacts of matrix habitat loss on native biota be considered as part of regional planning strategies.

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1. Introduction

In many areas that have long been cleared of most native vegetation for agriculture, relatively recent trends towards declines in biodiversity are becoming evident. Large-scale shifts to intensive land uses in regions long used for traditional, more extensive agriculture have been implicated in such declines, particularly in Europe (Fuller et al., 1995; Chamberlain et al., 2000; Newton, 2004; Eggleton et al., 2005). The ever-increasing pressure for greater production efficiency in farming systems is continuing to drive the trend toward more intensive agricultural practices (Mansergh et al., 2006).

At the scale of the individual paddock, agricultural intensification involves a simplification of the agroecosystem through reduced biodiversity, and increased inputs in the form of pesticides, fertilisers, or water (Tscharntke et al., 2005). Such intensification results in the removal of structural
elements of the matrix, for example the removal of paddock trees (Maron, 2005), as well as resulting in direct mortality for species susceptible to chemical inputs. The impacts of such loss and alteration of matrix habitat are evident at both the paddock and the landscape scale (Eggleton et al., 2005; Tscharntke et al., 2005). Previous studies have demonstrated that matrix quality can affect the ability of species to traverse the landscape by influencing the effectiveness of stepping stones and corridors (Baum et al., 2004), and paddock trees in particular have been shown to provide nesting and feeding habitat (Law et al., 2000; Lumsden et al., 2002; Manning et al., 2004; Lumsden and Bennett, 2005) and act as stepping stones between native vegetation remnants (Fischer and Lindenmayer, 2002). Furthermore, they directly and indirectly fulfill important ecological functions in influencing soil nutrient and moisture levels and harbouring beneficial species such as predatory invertebrates (Wilson, 2002; Oliver et al., 2006).

As climate becomes increasingly variable and rainfall in many temperate and semiarid agricultural regions becomes less reliable, reliance on irrigation of crops, even in dryland farming regions, is likely to increase. Centre pivot and lateral move irrigation systems, which pump water through a spray arm that can be over 600 m long, require the removal of all tall native vegetation within the reach of the arm. Despite only becoming widely used in many countries within the past 15–20 years, it is now one of the major forms of irrigation in suitable regions and its use is rapidly increasing. For example, in Mauritius the area under centre pivots reached 3000 ha within six years of the introduction of centre pivot technology (Teeluk, 1997). Where this proliferation of centre pivot irrigation replaces less intensive dryland farming practices, the nature of the matrix habitat is significantly affected, and there is potential for a large-scale impact on regional biodiversity.

We used an agricultural region of south-eastern Australia as a case study to investigate the extent of agricultural intensification, namely, introduction of centre pivot irrigation systems over a 23-year period. The proliferation of centre pivot irrigation circles is of concern in this region as the matrix areas still being farmed using less intensive grazing or dryland cropping support paddock trees which are an important food resource of an endangered taxon, the south-easteren red-tailed black-cockatoo (Calyptorhynchus banksii graptogyne), and habitat for several other threatened species (Maron, 2005; Maron et al., 2005). The rate of loss of these trees to 2005, previously only reported for a period early in the history of centre pivot use in the region (Maron, 2005), is also assessed.

2. Methods

2.1. Study area

An area of 163,200 ha in the western part of the Wimmera bio-region in Victoria was selected as a case study for this research (Fig. 1). The soils of the study area are primarily fertile grey clays interspersed with low-fertility sandy ridges, with native vegetation on the former soil types substantially modified or removed (Land Conservation Council, 1985). Within this study area, five focal landscapes were chosen to determine loss of scattered buloke (Allocasuarina luehmannii) trees (Maron, 2005), which are of particular interest due to their importance to the endangered red-tailed black-cockatoo (Maron and Lill, 2004) (Fig. 1).

2.2. Increase in centre pivots

The number and area of centre pivot irrigation areas (centre pivots) was determined from analysis of satellite imagery and aerial photography for the years 1980, 1993, 1995, 2000, 2001 and 2005. Landsat MSS imagery was used for the 1980 analysis (50 m pixels), Landsat TM imagery (30 m pixels) was used for 1993 and 1995 analysis, Landsat 7 EMR+ imagery (30 m pixels) was used for the 2000 analysis, SPOT Panchromatic/Monochromatic imagery (10 m pixels) was used for the 2001 analysis and ortho-rectified aerial photographs (0.6 m pixels) for the 2005 analysis. All active pivots and evidence of previous pivots were first calculated from the 2005 images as they provided the best resolution. The mapped areas were then compared to images from the previous years and any pivots that were not evident at an earlier time deleted. Due to the lower resolution of images earlier than 2001 it was not possible to determine which pivot areas were in use at the time. Thus the calculations are cumulative and represent the total area that had been used for centre pivots over the period. Although the satellite imagery and aerial photography were gathered at different times of the year and often over a time-frame of up to 12 months, for the purposes of calculating annual increase in pivot numbers and area the data were treated as though they were collected at the same time of the year. The area affected by centre pivots in 2005 in relation to the modelled pre-1750 distribution of ecological vegetation classes was calculated within ArcView GIS 3.3.

2.3. Loss of paddock trees

Rates of loss of paddock buloke trees between 1997–2005 were determined using the five focal landscapes. Following the methods of Maron (2005), all buloke paddock trees evident in each focal landscape were counted in the 2005 images and compared with the number present in 1997. However, although both sets of images used in this previous study were captured during summer, when grass residue was pale resulting in a high contrast between trees and pasture/crop stubbles, the 2005 images were captured at a time when the pasture grasses and crops were still green. This led to increased difficulty in distinguishing the trees in some images, particularly that depicting the Patyah landscape. In order to reduce error caused by omitting trees that were present but poorly visible, transparencies used to mark all trees in the 1997 image were overlaid on the 2005 images and each location where a tree was present in 1997 was individually checked for the presence of a tree in 2005. If there was uncertainty about whether a particular tree was present in 2005, it was considered to be present; thus, a conservative estimate of loss was made. Rates of loss from each landscape were compared with those recorded between 1981/82 and 1997 (Maron, 2005).
3. Results

3.1. Changes in land use

No evidence of centre pivots was visible on the 1980 images. However, several pivots were present in the area in 1993 and the number of and area affected by centre pivots increased steadily from 1993 to 2005 (Figs. 1 and 2). On average, 11.75 pivots totalling 617 ha were established per year from 1993 to 2005, although between 2000 and 2005, 13 new pivots at 651 ha were being established per year. Pivot areas proliferated in the north of the study area, particularly in the region between the Tallageira Nature Conservation Reserve and Little Desert National Park (Fig. 1). Few were established in the south. The average size of pivots increased slightly from 41.7 ha in 1993 to 50.5 ha in 2005. By 2005, 8734 ha of the study

Fig. 1 – (a) Location of study area and focal landscapes and distribution of areas affected by centre pivot irrigation in the west Wimmera in (b) 1993; (c) 2000 and (d) 2005 (note 2001 SPOT satellite imagery used as background). Pivot areas are represented as black circles. Black squares = focal landscapes.
area had been affected by centre pivots, representing 5.5% of the area and up to 25% of the area between Tallageira Nature Conservation Reserve and Little Desert National Park.

Areas which formerly supported the Plains Woodland ecological vegetation class were the most heavily utilised for centre pivots, making up almost 75% of the total area affected by pivots, while Plains Woodland, Plains Savannah and Shallow Sands Woodland ecological vegetation classes combined made up almost 94% (Table 1). All of these ecological vegetation classes have buloke as a dominant or co-dominant overstorey species (Commonwealth and Victorian Regional Forest Agreement Steering Committee, 2000; White et al., 2003).

3.2. Overall rates of tree loss

In the 15 years between 1981/82 and 1997, the average rate of loss of buloke trees from paddocks in the five landscapes analysed had been 26% (Maron, 2005). In the 23 years to 2005, this figure was 42%. In the most affected landscape, 70% of trees that were present in paddocks in 1981/82 had been lost by 2005 (Table 2).

The per annum rate of loss of paddock buloke trees, expressed as a percentage of trees present in 1981/82, was higher when calculated over the 23 years to 2005 than over the 15 years to 1997 (Table 3). While Maron (2005) found an average annual rate of loss of 1.7%, the rate of loss per annum since Table 1 – Area (ha) of pre-1750 ecological vegetation classes (EVC) affected by centre pivot irrigation areas within the study area

<table>
<thead>
<tr>
<th>Pre-1750 EVC</th>
<th>Total area under pivots in 2005</th>
<th>% of pre-1750 extent under pivots</th>
</tr>
</thead>
<tbody>
<tr>
<td>Damp sands herb-rich woodland</td>
<td>97.6</td>
<td>2.4</td>
</tr>
<tr>
<td>Heathy woodland</td>
<td>5.8</td>
<td>0.2</td>
</tr>
<tr>
<td>Low rises woodland</td>
<td>12.0</td>
<td>1.1</td>
</tr>
<tr>
<td>Lowan sands mallee</td>
<td>53.2</td>
<td>0.7</td>
</tr>
<tr>
<td>Dunefield heathland</td>
<td>0.4</td>
<td>1.0</td>
</tr>
<tr>
<td>Sandstone ridge shrubland</td>
<td>15.1</td>
<td>2.0</td>
</tr>
<tr>
<td>Heathy herb-rich woodland</td>
<td>4.6</td>
<td>0.1</td>
</tr>
<tr>
<td>Seasonally inundated shrubby woodland</td>
<td>2.3</td>
<td>0.7</td>
</tr>
<tr>
<td>Red gum swamp</td>
<td>49.5</td>
<td>0.6</td>
</tr>
<tr>
<td>Drainage-line woodland</td>
<td>4.6</td>
<td>0.6</td>
</tr>
<tr>
<td>Shallow sands woodland/</td>
<td>1.7</td>
<td>0.1</td>
</tr>
<tr>
<td>plains sedgy woodland/seasonally inundated shrubby woodland/damp sands herb-rich woodland Mosaic</td>
<td>284.0</td>
<td>14.2</td>
</tr>
<tr>
<td>Damp sands herb-rich woodland/shallow sands woodland mosaic</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plains woodland</td>
<td>6419.6</td>
<td>6.4</td>
</tr>
<tr>
<td>Plains savannah</td>
<td>838.1</td>
<td>14.6</td>
</tr>
<tr>
<td>Shallow sands woodland</td>
<td>803.6</td>
<td>6.3</td>
</tr>
</tbody>
</table>

Table 2 – Loss of paddock buloke trees since 1981/82 in the five focal landscapes

<table>
<thead>
<tr>
<th>Focal area name</th>
<th>Area (ha)</th>
<th>No. buloke trees 1981/82</th>
<th>No. buloke trees 2005</th>
<th>No. lost</th>
<th>% loss since 1981/82</th>
</tr>
</thead>
<tbody>
<tr>
<td>Neuarpurr</td>
<td>2320</td>
<td>1586</td>
<td>476</td>
<td>1110</td>
<td>70.0</td>
</tr>
<tr>
<td>Neuarpurr North</td>
<td>2060</td>
<td>1662</td>
<td>759</td>
<td>903</td>
<td>54.3</td>
</tr>
<tr>
<td>Bringalbert</td>
<td>1440</td>
<td>4257</td>
<td>3064</td>
<td>1193</td>
<td>28.0</td>
</tr>
<tr>
<td>Benayeo</td>
<td>1500</td>
<td>1554</td>
<td>837</td>
<td>717</td>
<td>46.1</td>
</tr>
<tr>
<td>Patyah</td>
<td>530</td>
<td>411</td>
<td>363</td>
<td>48</td>
<td>11.7</td>
</tr>
<tr>
<td>Mean ± SD</td>
<td>7850</td>
<td>9470</td>
<td>5499</td>
<td>2503</td>
<td>42.0 ± 10.2</td>
</tr>
</tbody>
</table>

a After Maron (2005).
1997 (calculated as a percentage of 1997 trees) was 3.0% (Table 3). This indicates that although the number of trees remaining in the landscape is decreasing, the number being removed annually is not, and therefore the annual percentage loss has more than doubled in comparison with the period 1981/82–1997.

3.3. Impact of land use on tree loss

New centre pivot irrigation systems that had not been present in 1997 were evident in three of the five landscapes in 2005. These developments contributed disproportionately to the loss of paddock buloke trees. Although these new pivots affected an average of 7.6% of these three landscapes, they accounted for an average 22% of the bulokes lost since 1997 (Table 4).

Due to the time of year at which the images were taken (spring) pasture could not always be reliably distinguished from cropland. However, the Neuarpurr and Neuarpurr North landscapes were predominantly cropland in 1997 and personal observations in the study area confirm that this remained the case in the five years to 2005. The Benayeo landscape in the five years to 2005 was observed to have become predominated by dryland cropping. The Patyah landscape consists of pasture with very little cropping, while the Bringalbert landscape hosts a mix of cropping and pasture. The mean overall loss of trees from the three cropping landscapes, at 56.8%, is substantially higher than the Bringalbert landscape in the five years to 2005 was observed to have been significantly reduced. The Benayeo landscape hosts a mix of cropping and pasture.

<table>
<thead>
<tr>
<th>Focal area name</th>
<th>% of 1981/82 trees lost per annum</th>
<th>% of 1997 trees lost per annum</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Calculated over 15 years to 1997</td>
<td>Calculated over 23 years to 2005</td>
</tr>
<tr>
<td>Neuarpurr</td>
<td>2.6</td>
<td>3.0</td>
</tr>
<tr>
<td>Neuarpurr North</td>
<td>2.2</td>
<td>2.4</td>
</tr>
<tr>
<td>Bringalbert</td>
<td>1.2</td>
<td>1.2</td>
</tr>
<tr>
<td>Benayeo</td>
<td>2.3</td>
<td>2.0</td>
</tr>
<tr>
<td>Patyah</td>
<td>0.3</td>
<td>0.5</td>
</tr>
<tr>
<td>Mean</td>
<td>1.7</td>
<td>1.8</td>
</tr>
</tbody>
</table>

a See Maron (2005).

4. Discussion

Agricultural intensification, rather than the conversion of new areas to agriculture, has been responsible for most of the increase in food production over the past 30 years (United Nations Food and Agriculture Organization, 2006). Such intensification has resulted in substantial landscape change, and the resultant negative impacts on native biota are more recently becoming evident (Donald et al., 2001; Söderström et al., 2003). Continuing growth of centre pivot irrigation is likely to result in markedly changed landscapes, with the removal of the majority of woody native vegetation from the agricultural matrix. In the current study, the area affected by centre pivots increased linearly to cover 5.5% of the study area over the 23 years to 2005. At a more localised scale, centre pivots occupied as much as 25% of the area between the Tallageira Nature Conservation Reserve and the Little Desert National Park. This trend appears to be more widespread, as similar increases in centre pivots were evident in adjoining parts of the Wimmera bioregion in South Australia.

Although the groundwater accessed in the study area has been fully allocated since 1996 there has been no reduction in the rate of increase in area affected by centre pivots because the pivot arms are frequently moved to new paddocks with the original site used for the centre pivot returned, temporarily or permanently, to dryland cropping or grazing (C. Guest, personal communication, 2006). Agricultural practices such as grazing and tillage prevent the regeneration of native vegetation on sites cleared for centre pivot irrigators. Therefore, although tree removal at one pivot site might involve only a few dozen trees, the cumulative impact as centre pivots are moved represents a substantial loss of matrix habitat on a regional scale.

This alteration in matrix habitat is likely to have had a substantial impact on the biodiversity of the region. Manning et al. (2006) refer to scattered paddock trees as ‘keystone structures’ due to their disproportionately large contribution to ecosystem function. In the Wimmera, the scattered buloke trees provide a critical food resource to an endangered specialist granivore, the south-eastern red-tailed black-cockatoo, which feeds only on the seeds of three tree species (Joseph, 1982; Maron and Lill, 2004). Scattered trees in agricultural land also act as focal foraging sites for microchiropteran bats (Lumsden and Bennett, 2005) and their role as host to mistletoes (including, in the case of buloke, the vulnerable buloke mistletoe Amyema linophylla) makes them likely nesting sites for a suite of declining bird species and sources of high-nutrient litter fall (Watson, 2001; Cooney et al., 2006). They also often represent the remnants of threatened vegetation communities (Gibbons and Boak, 2002).

In addition to local-scale habitat alteration, changes in the agricultural matrix can influence landscape-scale processes. The loss of paddock trees and other forms of matrix homogenisation, such as conversion from native perennial pastures to introduced annuals, results in an increase in patch/matrix contrast, potentially influencing use of remnant patches and decreasing the ability of some organisms to traverse the landscape. Lower genetic diversity in skink populations has been attributed to attributes of the matrix surrounding habitat patches, suggesting that skink dispersal
through a homogeneous exotic pasture matrix is reduced compared with a matrix of native tussock grassland (Berry et al., 2005). Castellón and Sieving (2006) recorded reduced dispersal of an understorey rainforest bird from wooden patches surrounded by open habitat than those surrounded by shrubby vegetation. Measures that manage matrix permeability may reduce the influence of fragmentation (Antongiovanni and Metzger, 2005). Improvements in matrix quality may be similarly effective at providing landscape connectivity as creation or retention of vegetation corridors (Castellón and Sieving, 2006).

The introduction of irrigation can itself have a negative effect on birds, especially those of open grassy habitats (Brotons et al., 2004). Irrigation and intensive management of areas previously subject to dryland agriculture results in changes to the spatial and temporal distribution of resources, through reduced prey habitat in the form of pasture or crop stubbles (McCracken and Tallowin, 2004). In Australia, bird species such as spotted harrier (Circus assimilis), little button-quail (Turnix velox), and plains-wanderer (Pedionomus torquatus), which are listed as threatened or near threatened at national or state levels (Department of Sustainability and Environment, 2003), are likely to be negatively affected by habitat change from previously open grassy woodland with extensive grazing and cereal cropping to intensive irrigated cropping.

The increase in centre pivot irrigation is effectively accelerating habitat loss in a region where tree decline through stubble burning, nutrification and natural senescence with little regeneration already are occurring (Maron, 2005). In an earlier study of the five focal areas, Maron (2005) recorded substantially lower annual rates of tree loss for the fifteen-year period to 1997 than we found in the eight years to 2005, despite native vegetation clearing controls being in place since 1989 with more recent policy intended to provide some protection for paddock trees introduced in 2002 (State of Victoria, 2002). The rates of tree loss recorded in this study are also higher than those recorded in other recent studies of agricultural regions (Ozolins et al., 2001; Carruthers, 2005).

For example, Carruthers (2005) reported an annual rate of loss of 0.5% compared to the average 1.8% over the 23 years of this study. If the 3.0% annual rate of loss detected over the eight years to 2005 continues, paddock bulokes will be all but absent from the area within approximately 25 years, although it is likely that trees in some landscapes (such as those in which the predominant land use is sheep grazing) will be retained longer.

The loss of paddock trees and other matrix habitat has not often been quantified in the past, with the focus mainly on broadscale clearing of intact native vegetation. Australian native vegetation legislation does not adequately provide for the loss of matrix habitat through intensification of land use, yet this is a major threat to biodiversity. The buloke grassy woodlands of the Wimmera are part of a vegetation community listed as threatened under the Commonwealth and State legislation. However, as this once widespread community is now restricted primarily to paddock trees within a matrix of pastoral and irrigated land with predominantly non-native understorey, the trees themselves are not afforded the level of protection they would receive if they occurred within a remnant patch.

In Europe and the United States, agri-environment schemes have been widely used in attempts to mitigate the negative impacts of agricultural intensification and restore matrix habitat (Donald and Evans, 2006). Primary producers are paid to employ more environmentally-friendly practices, which often include maintaining a more heterogeneous matrix through restoring field margins and reducing herbicide use. Should such schemes be introduced in Australia, compensation of land managers for extending matrix management to the protection of scattered paddock trees has the potential to create benefits for native wildlife on a landscape scale.

We recommend that consideration be given to the impacts of agricultural intensification on biodiversity in Australia. Although legislation to cease broad-scale clearing has been introduced in most states and territories, Australia potentially faces a second wave of local extinctions and species declines due to the continuing loss of matrix habitat. While centre pivot irrigation is preferable in terms of water efficiency compared with practices such as flood irrigation, the potential impacts on native biota of introducing it to previously non-irrigated areas warrant attention. Landscape planners in affected regions should consider activities to mitigate and offset the effects of matrix habitat change, potentially through retaining and replanting paddock trees wherever possible, and utilising the corners of centre pivot paddocks for revegetation.

Acknowledgements

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REFERENCES


Insights into the biodiversity and social benchmarking components of the Northern Australian fire management and carbon abatement programmes

By James Fitzsimons, Jeremy Russell-Smith, Glenn James, Tom Vigilante, Geoff Lipsett-Moore, Joe Morrison and Michael Looker

Summary  Much of northern Australia’s tropical savannas are subject to annual intense and extensive late dry season wildfires, much of this occurring on Aboriginal land. Based on the successful West Arnhem Land Fire Abatement (WALFA) model, which has resulted in significantly reduced greenhouse gas emissions, fire abatement programmes are planned for other significant regions of northern Australia. This study offers an introduction to the ideas behind a proposed environmental and social benchmarking project that aims to evaluate the potential benefits of expanding the fire abatement program in northern Australia, under the leadership of NAILSMA and its partners. Gaining a better understanding of the biodiversity, social and cultural outcomes of these fire abatement activities is an important component of demonstrating multiple benefits of these programmes. We emphasize the role of both biodiversity and cultural mapping to establish benchmarks and baseline states, with the involvement of Indigenous communities being a key element to optimize social and biodiversity benefits. Consultation with Traditional Owners and ranger groups to establish an agreed set of targets, indicators and sampling protocols and methodologies are critical component of this process. Examples of preliminary work to date are provided.

Key words: benchmarking, Indigenous fire regimes, Indigenous livelihoods, monitoring, tropical savannas.

Introduction

Currently, an average of 350 000 km² of Australia’s 1.9 million km² sparsely settled northern savannas are burnt annually, mostly by relatively intense and extensive late dry season wildfires. Following widespread disruption to customary Indigenous (Aboriginal) fire management practices associated with European settlement, contemporary savanna fire regimes are recognized as having globally significant impacts on biodiversity (e.g. Woinarski et al. 2011), soil conservation (Townsend & Douglas 2000; Russell-Smith et al. 2006) and emissions of greenhouse gases (Russell-Smith et al. 2009). Developing economically viable solutions for implementing ecologically sustainable landscape-scale fire management for these vast, remote, biodiversity-rich regions is a significant challenge. This task is founded on the rich and diverse Indigenous cultural landscapes of the north and growing appreciation for the knowledge and skill of Indigenous land owners and managers. Indigenous knowledge of fire in the landscape and how to manage it is embedded in complex customary relationships and a site-based ontology (James 2009). Despite relative success with what some have called the ‘two tool box’ approach to land and fire management (i.e. using customary and modern approaches), there is limited understanding by non-Indigenous people about Indigenous fire knowledge across the north (but see Yibarbuk et al. 2001; Russell-Smith et al. 2009). This is part of the challenge being taken up by the North Australian Indigenous Land and Sea Management Alliance (NAILSMA), ranger groups and project partners (Kimberley Land Council, Northern Land Council, Carpentaria Land Council Aboriginal Corporation and Balkanu Cape York Development Corporation), with other partner organizations from the science and conservation sectors. This builds on foundational efforts by Indigenous groups, government agencies and scientists in some of these northern Australia regions (e.g. Russell-Smith et al. 2009).

In this paper, we outline the biodiversity and social benchmarking components of the proposed monitoring for the broader Northern Australian Fire Abatement Program, an initiative that is still
under development. By 'benchmarking', we refer to the process of identifying common standards or criteria by which future qualitative and quantitative monitoring could measure the success or otherwise of six proposed regional fire management projects, as well as comparing changes over time within each case. As the 'benchmarking' programme is only in its early stages and the final design is yet to be fully developed, we do not present here the details of the experimental design of the biodiversity or social benefits monitoring or the ways the monitoring and evaluation will evaluate implications. Nor do we, owing to space constraints, address in detail here other important components of that programme such as governance arrangements and the carbon economy.

The West Arnhem Land Fire Abatement (WALFA) project

The West Arnhem Land Fire Abatement (WALFA) project has been developing since 1996 to address chronic fire management problems in Aboriginal-owned, high-biodiversity savanna landscapes of western Arnhem Land. In particular, the essential problem has involved depopulated landscapes and a subsequent increase in annual wildfires occurring late in the 7-month dry season period. Prior to European settlement, Aboriginal people burnt the landscape throughout the year, but with a strong emphasis on early- to mid-dry season burning (Garde et al. 2009). For example, over the 10-year pre-WALFA project baseline period, an average of ~40% of the 28 000 km² WALFA region was burnt annually, with 32% of this annual average occurring in the late dry season. Nearly the entire amount of this burning has been attributed to human (anthropogenic) ignitions, although lightening was also a factor (Russell-Smith et al. 2009).

To reduce the emissions of greenhouse gases, the key objective of the WALFA project has been to re-establish customary Indigenous fire management regimes, particularly to increase the extent of early season burning using strategically prescribed fires. It was considered that such a focus would enable the WALFA project partners to manage for and limit the extent of late season fires and thereby reduce both the area and amount of fuels which are burnt. In the first 6 years of operation (2005–2010) of the project, WALFA partners have reduced greenhouse gas emissions by ~50% relative to the established 10-year (1995–2004) pre-project baseline (WAFMA project reporting, unpublished data).

This research led to a landmark greenhouse gas offset agreement between ConocoPhillips, the Northern Territory Government, Northern Land Council, and Traditional Owners and Indigenous land managers in west Arnhem Land. This agreement recognized that significant greenhouse gas abatement could be achieved through savanna fire management carried out by Indigenous ranger groups as an offset to some of the greenhouse gas emissions generated at ConocoPhillips’ liquefied natural gas plant in Darwin Harbour. Under the arrangement, around $1 million a year is paid into the WALFA project for 17 years to provide this fire management service.

Importantly, the WALFA project provides first, extensive employment for Indigenous people in an environment where if any other culturally appropriate, non-government subsidised opportunities exist, and second, a long-term contractual foundation for other forms of natural and cultural resource management and for associated community and regional infrastructure development (Whitehead et al. 2009).

Building on 'WALFA' across the north

Building on the success of WALFA, the NAILSMA partnership has identified a number of other high-priority areas which would significantly benefit from the restoration of customary burning regimes. These regions are the North Kimberley, central Arnhem Land, the Gulf of Carpentaria, western Cape York, and the Daly River-Port Keats area (Fig. 1). The intention is not to replicate all aspects of WALFA. For example, governance arrangements for the fire programmes are likely to vary because of the differences in culture, land ownership, tenure and legal environments. In many areas, especially where legal land tenure arrangements have enabled good access, local Indigenous people remain or have re-engaged in fire management (e.g. Fisher et al. 2004). There is significant extant customary knowledge and skill in fire management in regions outside of WALFA, although its deployment and coverage is limited owing to a range of
current social factors including, among other things: limited visitation opportunities with a history of depopulation from and disenfranchisement of ancestral land; lack of requisite familial cooperation; lack of operational resources; and, poor land access infrastructure. Further, there are lessons from the WALFA experience which would also inform the establishment of fire programmes in other regions. For example, Lendrum’s (2007a,b) assessment of WALFA in relation the various benefits achieved was that the market driven resource management approach had prioritized the environmental and economic outcomes over the social and cultural outcomes.

Need for biodiversity and social benchmarking

There has been significant investment in fire and greenhouse gas emissions science in the WALFA region (Russell-Smith et al. 2009), which has enabled accurate assessments of amounts of greenhouse gas abatement from changes in fire regimes. However, just as importantly there are also likely to be important biodiversity benefits arising from implementing more traditional fire regimes (such as more heterogeneous habitat for fauna at a landscape scale; more food resources; e.g., Fitzsimons et al. 2010; Radford 2010), as well as employment, social and cultural benefits for Indigenous people (including improved health by working on country; e.g., Burgess et al. 2009; Garnett et al. 2009; Weir et al. 2011). In addition to current voluntary (as per the WALFA agreement) and mooted Australian regulatory carbon offset opportunities, such additional biodiversity and social benefits are likely to be especially attractive for potential investors, including the philanthropic sector. However, to be able to refer with any certainty to benefits in these fields, internationally accredited and recognized benchmarking and monitoring programmes are essential. In this case we are seeking to measure the performance and impact of fire abatement projects against criteria that are in the process of being established (i) by the Indigenous land and project owner group and (ii) by the regulatory and market environments into which land managers will seek to trade the products of their work and expertise.

This initial biodiversity and social benchmarking project being undertaken through the NAILSMA partnership is largely field based. It is built on maximizing engagement with and building the capacity of Indigenous landowners in collaboration with management, technical, and research expertise from a range of institutional partners across northern Australia. Key components are outlined below.

Foundation mapping activities

For both biodiversity and social foundational mapping activities, a combination of desktop data collection, field-based participatory planning, modelling and field testing will typically be undertaken during the benchmarking and monitoring programme. The integration of biodiversity and social benchmarking is an innovative and important element of this project. Understanding the specific linkages between biodiversity conservation and social and cultural values in northern Australian savannas is critical for the development of economically and ecologically sustainable natural resource management and self-determination outcomes over the longer term (see Schmidt & Peterson 2009 on developing countries).

Biodiversity

Vegetation, biodiversity and fire mapping layers and tools are fundamental to undertaking the benchmarking and monitoring project, as well as contributing more broadly to natural resource management planning across the north. For each project area, a first task is to develop detailed vegetation structure and fuels mapping, where these do not already exist, as a core input into the savanna burning emissions accounting methodology. For biodiversity benchmarking purposes that mapping needs to be further refined to map habitats, key seasonal water sources, ongoing mapping of fauna and flora records, habitat condition, and where appropriate, sites and assets of cultural significance (e.g. Russell-Smith et al. 2009). In most cases, this is likely to use a combination of techniques and approaches. For example, the methods and techniques used for vegetation mapping may be driven more by western science and undertaken by research institutions, but with the involvement of Indigenous rangers and Traditional Owners in site selection and field validation being an important component. On the other hand, biodiversity assets (such as species and habitats) which are culturally significant will be identified by Indigenous rangers and Traditional Owners as part of
participatory planning and form an important component of planned long-term monitoring activities (see examples from the Kimberley outlined below). From this, ‘benchmark states’ will be identified against which future monitoring of the condition of the landscapes will be measured. Evaluation of improvement from current states will be informed by a monitoring plan.

Social and cultural

Local Indigenous ownership of fire/emissions abatement projects is considered to be an important element in the success of projects. This means that the degree of actual and perceived ownership is, in its own right, an important benchmark against which to evaluate the success of future projects. Identifying other benchmarks of relevance to local Indigenous owners therefore requires project developers to capture local values and aspirations in a participatory process with owners, bridging cultural differences.

Another identified benchmark is the extent to which customary knowledge is deployed in the future projects. The baseline state is that there is much extant customary knowledge of fire and related skills that is not currently being deployed in many parts of northern Australia. Local Indigenous people are keen to reconstruct customary relationships, local languages, protocols and means for cooperation. A basic premise of the fire management project is that Indigenous stewardship of the on-ground programme can provide such a purposeful context. Altman (2009) makes this point more broadly on Indigenous environmental stewardship and cultural diversity. Local cultural mapping activities manifest land managers revitalizing their connection to country (e.g. Moorcroft et al. 2012). These activities are being undertaken by various groups to, for example: (i) help clarify who ‘speaks for’ which land area, (ii) get public consideration for sacred and significant site locations, (iii) negotiate boundaries and encourage cooperative planning amongst families, (iv) provide instruction for young people about the many layers of meaning in the customary landscape including ancestral creation, and (v) plan for multiple land use interests. Although a common ethnographic tool, this type of mapping is in its early stages as a formal process in the fire project. Mapping in this sense may not be a public or open process, often involving sensitive issues such as: local interpretations of ancestral law; familial conflicts; gaps in local knowledge; leadership; and, claims to ancestral country. Understanding and addressing sociocultural variables as expressed in these activities, often influenced by historical circumstance, is challenging but important in strengthening foundational governance (Cernea 1991).

Integrating biodiversity and cultural mapping

The processes described above show that mapping tools – whether cultural or biodiversity focused – can not only be useful in identifying benchmark indicators but are also useful in describing the baseline state and trends for social/cultural attributes. Baseline information mapping may also include land use and interest, tenure type/extent, cultural and economic resource values, demography, infrastructure, and cultural and heritage site identification and condition where change and impact can be visually tracked over time. The mapping of stakeholders’ current and potential land use interests, social, cultural and economic resource values, and institutional arrangements will be important to develop baseline information for (i) monitoring the impact of the project in the long-term, (ii) informing the direction and management of the project, and (iii) gaining a comprehensive understanding of all interests and values attached to the resource. Mapping can also be a useful communications and management tool in this challenging management space at the interface of customary interests and market engagement (see Tobias 2009).

Benchmarking activities – proposed project design

A range of benchmarking activities focused on social and biodiversity issues will be brought together and synthesized across the six proposed regional projects. A key
The benchmarking component will be the integration of social and biodiversity benchmarking activities with the involvement of Indigenous groups in both cultural and biodiversity mapping processes.

The lessons that will be learnt through these early phase assessments are likely to have far-reaching implications for: (i) the sustainability of Indigenous-owned natural resource management enterprises; (ii) the integration of scientific and sociocultural processes in natural resource management research methodologies and project design and management; (iii) the undertaking, assessment and auditing of Australian national cultural and natural resource management programmes and activities; and (iv) the testing of evidence-based criteria and frameworks for informing development of ‘biodiversity credits’ and ‘social and cultural credits’ associated with savanna burning. These ‘credits’ may be described as units of monetary value credited to a project proponent for measurable amounts of benefits (in this case biodiversity and social and cultural) that are delivered by the project or set of activities as additional to any benefits that may be created under normal circumstances (i.e. without that project). Realizing income from credits for delivering measurable beneficial outcomes depends on there being a market mechanism in place.

The biodiversity benchmarking component will specifically address:

- efficacy and appropriateness of selected biodiversity indicators, condition targets, and associated assessment regimes for landscape-scale benchmarking;
- critical cost-benefit assessments of management activities to guide ongoing development, refinement and investment in savanna biodiversity management; and
- exploration of international standards, frameworks and investment opportunities concerning ‘biodiversity credits’

The social benchmarking component is at an early stage and has adopted a bottom-up approach to address project monitoring and evaluation requirements for customary land owners and managers, and monitoring and evaluation requirements for enterprise level engagement with external partners, regulators and markets. This is being developed initially through field-based participatory activities with land owners and managers aimed at:

- eliciting values and aspirations held by customary land owners and managers in land and fire management, with careful reference to income potential and enterprise qualities of the projects (e.g. equity and resource distribution);
- developing these into project targets, relevant indicators, monitoring, reporting and project management strategies for adaptive management; and
- accountability to the customary land owners and managers

These activities are complemented by a desktop and institutional level participatory research approach to include targets and measures relating to co-benefits assessments, market regulators and relevant industry standards accreditation. For example, monitoring and evaluation of social benefits may be required to achieve premium ‘Indigenous Carbon’ credits under Australia’s Carbon Farming Initiative (http://www.climatechange.gov.au/cfi) or registration with the International Climate, Community and Biodiversity Alliance Standard (http://www.climate-standards.org/) – so the terms required for these various accreditations will be considered in the initial design.

The benchmarking and monitoring process aims to complement existing community-based management approaches with practical tools for enhancing the efficacy and sustainability of Indigenous project ownership and collaborative management. Significant questions about the costs, relative stages of regional project development and differences in governance arrangements have been raised by land management groups and their representative organizations across the project as a whole (NAILSMA 2010a). To help address such issues, relevant to this northern savanna-scale market opportunity, critical cost-benefit assessments to guide regional and trans-regional development, investment in social capital and operational capacity are being discussed. The benchmarking programme will help capture this and provide an important description of development and
and change to assess against baseline values and aspirations.

Field consultations have confirmed the need for an ongoing ‘two way’ communications strategy and the development of local and project scale management tools and governance arrangements. The benchmarking process will serve these two purposes (see also Taylor 2004 on regional planning). It is anticipated that at the local level, articulating key relevant values in cultural and natural resource management, identifying practical indicators, and establishing monitoring and project management accountability will enhance Indigenous project ownership and build capacity to govern (see also Ens et al. 2012).

It is recognized that efforts to integrate socioeconomic and biophysical data in natural resource management can be challenging (e.g. Curtis et al. 2005), and there are many complex elements to the benchmarking process outlined above. Whilst this programme is unique there are challenges identified elsewhere that are of relevance here. For example, the geographic size and shape of project regions will not be the same for biodiversity interests as they might be for Indigenous interests. Defining (conceptually and geographically) discrete project regions will impact the measurability of indicators across all project regions (Taylor 2004). Further, the diversity of Indigenous languages raises issues of continuity of meaning and data collection over the whole region (see also Buzzard 1990 on social analysis in developing countries) manifesting the broader issue of multiple perspectives and goals in the relationship between Indigenous land management and biodiversity conservation; a self-determination issue for land managers and policy makers worldwide (Schmidt & Peterson 2009). There are also logistical, financial, policy and seasonal complexities over this multi-jurisdictional northern savanna region.

With Indigenous people increasingly articulating the need to keep track of (contemporary) fire management impacts, their engagement in the fledgling benchmarking process has been strong and should allow longer field testing time and scope to review and deal practically with the challenges ahead.

**Activities to date and future directions**

There is an identified need for a programme of regional on-country workshops as a starting point for the benchmarking programme. A key component of this is consultation with Traditional Owners and ranger programmes to establish an agreed set of targets and indicators is a high priority along with a set of sampling protocols and methodologies that are culturally appropriate.

Most biodiversity benchmarking to date has focused on vegetation mapping at various scales. In the Kimberley, fire history and structural vegetation mapping has been completed, while fine scale habitat mapping is currently underway in the Mitchell Plateau and Pantijan (a collaboration between NAILSMA, the Kimberley Land Council, the Western Australian Department of Environment and Conservation, The Nature Conservancy, Curtin University, ranger groups and Traditional Owners). On the Arnhem Plateau, mapping of fire-sensitive *Allosyncarpia ternata* rainforest patches has been completed, and an assessment of their condition is underway in association with mapping projects for upland wetlands (a collaboration between Charles Darwin University, NAILSMA, The Nature Conservancy, and Manwuruk Rangers).

A workshop that included field activities, held in September 2010 at the Ngaliyindi Outstation in north-east Arnhem Land, sought to progress biodiversity and social benchmarking in the Central Arnhem Land Fire Abatement area. The workshop was hosted by Gurrwuliwing Rangers, SE Arfura Rangers and the people of Ngaliyindi and focused on the remote Arfura Swamp area, which has high biodiversity and cultural values (Weston et al. 2012). While only preliminary, activities undertaken in this area by Traditional Owners, rangers and other experts include local surveys of the condition of fire-sensitive rainforest patches and Northern Cypress Pine (*Callitris intratropica*) stands and three nights’ trapping for small mammals associated with a perennial rainforest spring system close to the Ngaliyindi Outstation (NAILSMA 2010b).

At the North Kimberley site, a large on-country workshop was held at Hann Gorge, Gibb River Station, in July 2010 bringing together Traditional Owners and Indigenous rangers from four large native title claim areas to discuss the principles of biodiversity and social benchmarking. Local monitoring was introduced by NAILSMA and partners through semi-structured interviews recorded using voice recorders and later turned into multimedia products together with photographs.

Fauna such as macropods (e.g. *Macropus spp.*) and Emu (*Dromaius novaebollandiae*) along with edible fruit-bearing trees have been identified as culturally important species whose health is linked to fire regimes (e.g. Vigilante & Bowman 2004; Vigilante et al. 2009). This was also articulated by Traditional Owners in the Wunambal Gaambera Healthy Country Plan (Wunambal Gaambera Aboriginal Corporation 2010, Moorcroft et al. 2012) and at the Hann Gorge workshop. As part of the vegetation mapping field programme at the Mitchell Plateau and Pantijan, Traditional Owners and Indigenous rangers, with the assistance of the Kimberley Land Council’s Land and Sea Unit, have begun to identify and document the locations of culturally important species and some initial long-term monitoring points are being established and methods trialled for these species in relevant habitats. This has included vehicle-based transects for macropods and Emu (Fig. 2) and plot-based monitoring of fruit tree species starting with *Buchanania* and *Vitex* species (Figs 3 and 4).

The benchmarking programme is still a work in progress, and there is still much consultation and ‘bottom-up’ planning to be conducted. Over the next couple of years, we will be seeking investment buy-in for the various northern Australian fire project areas. Achieving standardized baseline sampling across all fire programme areas is a high priority to feed into monitoring, reporting and verification of carbon agreements across the north.

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References


Ecological connectivity or Barrier Fence? Critical choices on the agricultural margins of Western Australia

By Keith Bradby, James A. Fitzsimons, Andrew Del Marco, Don A. Driscoll, Euan G. Ritchie, Jenny Lau, Corey J. A. Bradshaw and Richard J. Hobbs

Western Australia’s State Barrier Fence represents a continuation of colonial era attitudes that considered kangaroos, emus and dingoes as vermin. Recent plans to upgrade and extend the Barrier Fence have shown little regard for ecological impacts or statutory environmental assessment processes.

Key words: barrier fences, corridors, Dingo, ecological connectivity, Emu.

Great Western Woodlands.

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Introduction

Fragmentation of primary habitats is one of the major drivers of species’ declines and extinctions (Fahrig 2003; Fischer & Lindenmayer 2007; Gibson et al. 2013). Fragmentation occurs when remnant habitat is left isolated and the matrix is converted to other uses or from the construction of barriers such as fences or roads that impede the movement of individuals (e.g. Forman et al. 2003; Krausman & Harris 2011). As such, removing barriers and restoring destroyed or degraded matrix to increase habitat connectivity represents one of the most tractable ways to improve population viability over the long term (Lindenmayer et al. 2008). The increasing role in conservation programmes of connectivity conservation, through what are often popularly called ‘biodiversity corridors’, to reconnect...
landscapes (Soulé & Terborgh 1999; Fitzsimons et al. 2013) therefore represents an important progression from earlier focuses on protected areas that by themselves are not sufficient to retain biodiversity (e.g. Laurence et al. 2012).

Connectivity conservation recognises the need for integration and cooperation across different land uses (Worboys et al. 2010; Fitzsimons et al. 2013). There has been a rapid growth in the development and implementation of on-ground connectivity conservation initiatives in Australia, ranging from large subcontinental corridors to more regional initiatives (Fitzsimons et al. 2013). Despite this enthusiasm and the successes achieved, programmes to increase ecological connectivity inevitably come up against proposals, which if implemented, are likely to reduce connectivity.

In this review, we describe one such situation where hard-won conservation gains are now being threatened by a proposal to extend and upgrade an existing barrier fence designed to restrict Emu (Dromaius novaehollandiae) and Dingo (Canis dingo) distributions (Figure 1). The Government of Western Australia proposes to extend the existing fence some 650 km, from 1170 km to >1870 km. We contend this will seriously constrain terrestrial native mammal and Emu movement between south-western Australia and the rest of the continent and have wider ecological consequences. The proposed extension would also bisect the eastern end of Gondwana Link (Figure 2), the first large connectivity programme established in Australia (Bradby 2013) and recognised as one of six ‘foundation stone’ corridor endeavours in Australia (Australian Government 2012). We first describe the history of wildlife management using barrier fences in Western Australia, discuss their likely ecological impacts and then outline the process that has led to the planned extensions being considered without a full understanding of their negative ecological impacts.

**Western Australia’s barrier fence network**

Western Australia’s barrier fence network was initially established to prevent rabbits, introduced into eastern Australia and rapidly spreading westward, from reaching the main farming and grazing areas of Western Australia. The first fence – No 1 Rabbit Proof Fence – was constructed between 1902 and 1907 from the south coast to the Pilbara coast north of Port Hedland (Crawford 1967). Even before construction was complete, rabbits had breached the barrier and in 1904 work began farther west on a No 2 Fence (Figure 3).

Major deforestation across south-western Australia, largely to establish broad scale wheat growing, accelerated after 1918 (Jasper 1984; Beresford et al. 2001; Bradshaw 2012). Sections of the fence prevented Emus from trampling some, but not all, of the new wheat crops. In 1932, the Commonwealth Government tried unsuccessfully to ‘control’ Emus by engaging military personnel from the Fifth Military District using Lewis Machine Guns. This became known as the ‘Emu Wars’ and has received national derision (Marshall 1966; Serventy & Whittell 1967; Johnson 2006).

The fences were maintained, somewhat haphazardly, until the early 1950s, when they became officially known as ‘Vermin Barrier Fences’. Their effectiveness as a barrier against migrating Emus led to the addition of sections around the north-eastern Wheatbelt in the late 1950s and early 1960s, on the advice
of the Western Australian Government’s ‘Emu and Grasshopper Advisory Committee’ (Crawford 1967). Expansion of the Wheatbelt during the 1950s and 1960s (Jasper 1984; Beresford et al. 2001; Bradshaw 2012) made No 2 Fence largely redundant, and farms were established beyond the central and southern sections of No 1 Fence in the 1960s. Continued expansion of agriculture eastward was planned, but stalled in 1969 following a series of dry years and a global wheat surplus (Beresford et al. 2001). Further expansion became subject to a moratorium in 1985 following increased public concern about the agricultural marginality and high environmental values of areas being allocated for agriculture (Beresford et al. 2001).

Today, while the fence has remained largely intact in its northern sections, southern sections adjoining what is now known as the ‘Great Western Woodlands’ only exist in portions. From the 1980s onwards, these have been maintained as a barrier against Emu and Dingo movement onto farmland. However, trapping, poisoning and shooting Dingoes continued as a supplementary control measure, often on both sides of the fence.

Negative ecological impacts

The barrier fence network originally consisted of rabbit-proof mesh (mesh size ~3 cm across). This has now been largely replaced with 10-line fabricated netting (mesh size ~18 × 12 cm) with 1–2 wires, often barbed, along the top and a mesh ‘lap wire’, again of fabricated netting, buried in the ground and extending 450 mm from the fence. A 10-m cleared access track is maintained on either side of the fence (GHD 2012). Native vegetation on public land within 1–200 m of one side of the fence is often cleared of trees and shrubs by ‘chaining’ (a heavy chain dragged between two widely spaced bulldozers) followed by occasional burning to reduce flammability (GHD 2012). This vegetation clearing, accompanied by installation of access and maintenance roads for the length of the fence, is likely to be a barrier for the smaller species for which the fence itself is not a barrier (Brooker et al. 1999). A ‘buffer zone’ that can extend more than 15 km on either side of the fence is subject to an intensive Dingo and feral dog eradication programme using baiting, trapping and shooting (DAFWA undated) which enhances the barrier effect. Thus, the ecological impacts of the State Barrier Fence and associated works to control wildlife are likely to go far beyond the fence line.

While some research has been published to quantify the ecological impacts of fences in Australia (e.g. Pople et al. 2000; Somers & Hayward 2012) and elsewhere (e.g. Flesch et al. 2010; Lasky et al. 2011; Woodroffe et al. 2014), the potential impacts of the Western Australia barrier fence on native ecosystems are poorly studied. We therefore provide an evidence-based review of the likely or plausible impacts of the barrier fence and its proposed extension.

Loss of connectivity

Some porosity is inherent across almost any fence. Apart from Emus, most birds can fly over fences and reptiles can generally climb through or over the mesh, as can mammals below a certain size (depending on the mesh type used). Western Australia’s barrier fences are specifically designed to restrict the movement of large macropods, particularly Western Grey Kangaroo (Macropus fuliginosus), Red Kangaroo (M. rufus), Emus and Dingoes. Given the mesh sizes used, medium-sized macropods, such as Black-gloved Wallaby (M. irma) and other smaller wallabies surviving in remnant populations, are also unlikely to move across the existing fence and any future extensions unless it is breached by floods or windstorms. The fence might also impede the movement of Short-beaked Echidna (Tachyglossus aculeatus). Fence impacts on these and even smaller mammals have not been measured. For species such as the Black-gloved Wallaby, where long-term persistence depends on access to sufficiently large patches of suitable habitat (Courtney 1994; Short & Parsons 2004), reduced access to patches of available habitat could increase the risk of localised extinctions of populations. Given its design as an impermeable barrier to larger wildlife, it is likely that the fence and its extensions also increase the mortality of wildlife fleeing large wildfires.

Little is known of the fence’s wider ecological ramifications, such as disruption to the long-range movement of the target species, which is important for maintaining genetic adaptability particularly in light of climate change (Frankham et al. 2014). The possible exception is with Emu populations, about which more is known. URS (2007, p. 5-1) in their Benefit-cost Analysis of the State Barrier Fence noted that ‘… there is a definite movement of Emus southward in winter and northward in summer. At this time, Emus will travel up to 1000 km’. When these southward-moving Emus reach the Barrier Fence, they have been shot, poisoned or left to starve in the tens of thousands (APB 2001; Johnson 2006) (Figures 4 and 5). This problem is likely to be exacerbated by extending the fence.

Feral-proof fences have recently been established around a number of large private protected areas, usually to protect small mammals from predators such as feral Cat (Felis catus) and Red Fox (Vulpes vulpes) (e.g. Moseby et al. 2009). As with barrier fences these aim to keep particular species on one side of the fence, but there are many important differences. Feral proof fences target invasive species.
not native wildlife. They establish ‘enclosed islands’ around which species can move, so only negatively impact localised connectivity, and they can have ecological benefits. Barrier fences traverse large distances and are specifically designed to prevent wildlife movement at regional and continental scales.

**Damage to ecological functions**

*Restricting genetic transfer*

The Emu is an important seed disperser and can have strong influences on the diversity of vegetation by carrying many seeds long distances (Noble 1975; Calvino-Cancela et al. 2006; Dunstan et al. 2013). The germination of some seeds is also helped by their passage through the Emu’s gut (Noble 1975; Noble & Whalley 1978). Chalwell and Ladd (2005, p. 446) note that for many areas ‘...the restriction of the range of Emus as a result of agricultural development, a key seed disperser has been lost’.

*Trophic imbalance*

Barrier fences are generally justified by the extent to which they reduce the impact of predators (Dingo and feral Dog, *Canis familiaris*) and herbivores (Emu and kangaroos) on crop and livestock production (Letnic et al. 2011a). These have potential to be contradictory roles because the Dingo is effective in reducing the density of native herbivore populations that might otherwise increase and overgraze pastoral areas (Ritchie & Johnson 2009; Letnic et al. 2011a, 2012).

*Mesopredator release*

Barriers constructed to impede Dingo movement, which limit Dingo population growth and size on the agricultural side of the barrier, also lessen the important benefits of intact and functioning packs of Dingo in reducing feral Cat and Red Fox abundance. This in turn reduces the negative effects of these feral species on native wildlife, particularly on small to medium-sized mammals (Ritchie & Johnson 2009; Letnic et al. 2011a,

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**Figure 3.** Locations of the original Rabbit Proof Fences in 1907 and subsequent major extensions (adapted from Agriculture Protection Board 2001). Shaded area represents cleared areas.

**Figure 4.** Parent and chicks separated by the Western Australian State Barrier Fence on the Karroun Hill Nature Reserve boundary. (Photo Andrew Hobbs.)
This is unlikely to affect areas inland of the fence, although associated Dingo control programmes could be responsible for increasing feral predator abundance there.

Adaptability to climate change
Reducing wildlife connectivity, including that of Emus, inhibits the flow of genetic material, and further undermines the resilience of biota to adapt to climate change (Frankham et al. 2014). Additionally, the changes in vegetation structure predicted under conservative projections of climate change for this area (Schut et al. 2014) could make the perceived optimal location for any barrier likely to change.

Direct habitat loss
Each kilometre of the barrier fence (and associated roads) occupies a footprint of around 2 ha, based on a 20 m width (GHD 2012). Early evidence of high species richness, localised endemism and high heterogeneity in the plant species composition of vegetation communities and habitats across inland south-western Australia has been confirmed in recent years by Hopper and Gioia (2004). The proposed fence extension will result in a substantial footprint to accommodate its linear infrastructure, and this will further reduce already diminished habitat types and damage restricted and/or endangered plant communities and species, many of which have not yet been described.

Fragmentation and subsequent weed invasion of previously intact systems
The current barrier fence appears to have been responsible for only a few sporadic weed infestations, generally associated with camping areas and water points. However, it was constructed in an era when invasive weeds were less prevalent. With fence construction for the proposed extensions, ‘Weed invasion has the potential to be a serious issue’ (GHD 2012, p. 29), and new weed outbreaks have already resulted from contractors constructing access lines for government agencies elsewhere in the woodlands (Rob Trenordan, Granite-Woodlands Conservation Action Plan team, pers. comm.). Weed seeds are also likely to drop from machinery involved in ongoing maintenance of the proposed extensions and would be well positioned to colonise the disturbed areas associated with the fence, such as adjoining firebreaks and access tracks. Additional risk could exist from invasive Buffel Grass (Cenchrus ciliaris) and African Lovegrass (Eragrostis curvula), which are relatively recent arrivals and appear to be converging on the Great Western Woodland in particular a convergence likely to accelerate with climate change and lead to detrimental changes in fire regimes (Prober et al. 2012).

Little public consultation and science in the proposed barrier fence extension
Despite widespread concern about the ecological damage caused by barrier fences (e.g. CCWA 2012; Burton 2013) and the uncertainty over their economic value (e.g. Hayward & Kerley 2009; Flesch et al. 2010; Lasky et al. 2011), there has been an apparent reluctance at both political and departmental levels to engage meaningfully with these issues.

From publicly available documents, it appears that a Western Australian Department of Agriculture and Food (DAFWA) project to upgrade the current barrier fences was being developed by 2007 (URS 2007) and that between 2008 and 2010, political decisions were made to enclose south-western Australia behind one continuous barrier fence. The initial documented step was a broad cost–benefit study commissioned by DAFWA ‘...to investigate the potential of upgrading the SBF (State Barrier Fence) to a wild dog fence that would keep both emus and wild dogs out of the agricultural region’ (URS 2007, p. 8). This assessment considered the agricultural effectiveness of the barrier fence against three ‘target species’ – kangaroos, ‘wild dogs’ (i.e. the Dingo and feral Dog) and Emu. In terms of ecological issues and non-target wildlife species, consideration was limited to a brief, two-sentence statement that a staff member at the then Western Australia Department of Environment and Conservation ‘...suggests there is little adverse impact on non-target native species’ (URS 2007, p. 54). No other evidence to support this contention was provided.

The programme came to greater public attention in April 2010 when,
after it gained the support of Cabinet and funding through the Western Australia Royalties for Regions programme, the Ministers for Agriculture and Environment jointly announced the barrier fence upgrades and extensions, along with increased effort to bait and kill Dingoes and feral Dogs (Redman & Faragher 2010). The largest proposals within this programme are the construction of 150 km of new barrier fence between two existing sections known as the ‘Yilgarn Gap’ and some 650 km known as the ‘Esperance Extension’ that would extend the fence around the southern edge of the Great Western Woodlands (Redman & Faragher 2010; Grylls & Baston 2013) (Figure 1). Work began around 2010 on upgrades to the existing fence, with some 820 km upgraded to ‘wild dog standard’, a phrase used to describe improved effectiveness as a barrier against wildlife and feral dogs, and achieved primarily through the addition of mesh ‘lap’ wires where the fence meets the ground (DAFWA 2013a; Grylls & Baston 2013). Construction of the 165 km fence to fill the ‘Yilgarn Gap’ began in May 2014 (Baston & Redman 2014; DAFWA 2013b), with the project reportedly meeting Western Australia’s environmental and heritage approval standards (DAFWA 2012a), but without an assessment from the Western Australian Environmental Protection Authority. The final step in this programme is the major fence extension proposed for the Esperance area, which ‘aims to complete the physical barrier presented by the SBF (State Barrier Fence) from coast to coast and increase the resilience of vermin control in the associated agricultural areas’ (GHD 2012, p. 1).

The joint ministerial announcement of this proposed extension (Redman & Faragher 2010) was made without any comprehensive assessments of its likely ecological impacts or benefits, with the Department of Agriculture not undertaking an inter-

nal assessment until 2012 (Invasive Species Program DAFWA 2012). Even 2 years later, the then Department of Environment and Conservation were still ‘not conducting any research into the impact and effect of barrier fences as such’, even though they recognised their role as providing ‘the best advice we can into what the government’s proposal is’ (Standing Committee on Estimates & Financial Operations 2012, p. 25). Despite this, several assertions have been made in relation to the relative costs and benefits that are not supported by independent studies elsewhere.

For example, the then Minister for Environment stated, in support of the proposed fence, that ‘... wild dogs caused considerable damage to the environment, preyed on native wildlife and destroyed habitats’ (Redman & Faragher 2010). It was not specified how ‘habitats were being destroyed’ and this claim is contrary to extensive published literature (e.g. Ritchie & Johnson 2009; Letnic et al. 2012; Ritchie et al. 2012). Elsewhere, Department of Agriculture staff and farmer proponents have described the barrier fences as ‘non-lethal’ wildlife management tools (Esperance Express 2013; Read 2013), and this description gained some local currency (e.g. SCNRM 2013). Even discounting the tens of thousands of Emus regularly shot, poisoned or starved along the fence, many native animals, including kangaroos and wallabies, also clearly and obviously suffer and die when caught in the fence (Figure 6).

Political defence of the proposal to extend the State Barrier Fence continues in the face of the environmental problems outlined above, and despite there being no documented objective estimate of the impact on either target or non-target wildlife, or publicly available tests of the impact of the barrier fences on vertebrate fauna. This example from Western Australia corresponds with what is seen by many as a growing trend for governments in Australia to defend their land-use decisions by ignoring, dismissing or contradicting existing robust research on ecological impacts of particular activities (see, e.g. Fitzsimons 2012; Lindenmayer 2013).

Additionally, the extent of damage to agricultural productivity on the edge of the Wheatbelt caused by wildlife is poorly known, with DAFWA scientists lacking survey data and basing their conclusions largely on anecdotal evidence (URS 2007; DAFWA 2012b) that generally originates from the farmers who have been seeking the fence upgrades (Rampling 2011; Wynne 2011; DAFWA 2012a, 2013c, 2014) or from studies that include unconfirmed ben-

![Figure 6. Wildlife is regularly caught in the State Barrier Fence. This kangaroo had been trapped for many days when found on the Fence east of Lake Varley. (Photo Frank Rijavec.)](image-url)
benefits such as a presumed reduction in vehicle collisions with wildlife some 30–50 km from the State Barrier Fence (ERA 2009).

The intended placement and design of the Esperance component has become slightly less damaging since it became subject to wider public scrutiny. The initial concept plans were for a continuous fence of 450–490 km (URS 2007, p. 16; DRDL undated), sections of which were well inland of existing farmland and placed approximately 84,000 ha of the Great Western Woodlands into the agricultural zone (calculated using ARC-GIS by A. Keesing, Information Manager, Gondwana Link Ltd). Construction funds were allocated from the Western Australian Government’s Royalties for Regions programme and site works appeared imminent, with a 2015 deadline for completion (Redman 2010). Following adverse media coverage (e.g. ABC 2012), including an information programme run by private conservation groups (e.g. CCWA 2012) that focused on environmental impacts, DAFWA commissioned a scoping study to identify the ‘least constrained’ option for the route of the fence (GHD 2012), with current publicly available options now focused on a longer fence placed largely on farm boundaries (DAFWA 2013d), although this could change to reduce costs and with small ‘gaps’ in the fence where river valleys are to be crossed. DAFWA has subsequently commissioned flora, fauna and heritage surveys along the preferred route (DAFWA 2013d).

However, the information requested by DAFWA from its consultants (GHD 2012; DAFWA 2013d) focuses on site impacts, particularly the presence of species legally protected under Western Australian or national legislation. Broader considerations, such as disruption of large-scale genetic exchange, benefits of Dingoes for controlling feral predators and adverse impacts from other disrupted ecosystem services (e.g. seed dispersal) have had little consideration.

**Need for Public Environmental Review and a process to consider alternatives**

In Western Australia, proposals likely to have a large impact on the environment are ordinarily assessed by the Environmental Protection Authority (EPA), a statutory authority responsible for providing advice to the Minister for Environment under the *Environmental Protection Act 1986*. The Environmental Protection Authority applies a Public Environmental Review assessment where the proposal is of regional and/or statewide importance or has several key environmental factors or issues, some of which are complex or strategic (Office of the EPA 2012). The Western Australian Government’s programme of proposed upgrades and extensions to the barrier fence meets all or most of these criteria for assessment, and ideally, the Public Environmental Review process would have been applied to the entire programme of State Barrier Fence upgrades and extensions so that cumulative impacts could be assessed.

Despite the fence upgrades being underway since 2010, with detailed planning for the extensions underway since 2012, as of August 2014, the works and planned extensions had not been referred to DAFWA to the Environmental Protection Authority for formal environmental assessment, with the only formal referral for the Esperance Extension coming from the private conservation sector (CCWA 2013). From currently available documentation, it seems DAFWA is intending to submit a proposal for Environmental Protection Authority assessment following site-specific biological surveys on a single preferred fence alignment (DAFWA 2013d). Hence, it is likely that if a proposal is formally presented for statutory assessment, whether under the Western Australia legislative framework or nationally, the proposal will under state the impact on ecological processes across the landscape in which the fence will be constructed and maintained. It is therefore our view that an EPA assessment is needed and this should include the entire fence rather than sections. Considering the shortcomings outlined above a full benefit–cost analysis that considers alternative management approaches is also required and is likely to lead to better outcomes that are more acceptable to a broader range of stakeholders.

**Implications of inadequate consideration of environmental impacts**

Since 1902 when the State Barrier Fence was first established, our understanding of what species and ecosystems need for their long-term persistence has improved, along with a transformation in our understanding and appreciation of the biological richness of south-western Australia. This rapid and continuing increase of knowledge, combined with the area’s recent environmentally destructive history, has led to international recognition of the area as a global Biodiversity Hotspot ‘...where exceptional concentrations of endemic species are undergoing exceptional loss of habitat’ (Myers et al. 2000, p. 853).

This recognition reflects the area’s botanical richness (Myers et al. 2000; Hopper & Gioia 2004) and massive habitat loss (Bradshaw 2012). As such, many bird and mammal species that have mostly disappeared from smaller woodland and mallee remnants elsewhere in southern Australia are still relatively common in the area adjoining the proposed fence extensions, the Great Western Woodlands (Recher 2008; Bradby et al. 2011), which is large and intact enough to be the world’s largest remaining temperate woodland (Wat-
many grazing leases in the southern rangelands that adjoin the northern section of the existing fence are not commercially viable, largely due to declining terms of trade, reduced carrying capacity and reductions in palatable perennial shrubs available for stock (Government of Western Australia 2009). Many pastoral leases are now owned and managed for their conservation and cultural values or because they include important mineral resources (e.g. van Etten 2013; Fitzsimons et al. 2014); and the need for better landscape-scale planning to guide future land use is increasingly recognised (Safstrom & Waddell 2013).

The Western Australian Government is already proposing changes in tenure arrangements to support diversification of leasehold land, including for conservation use (WA Department of Lands 2013). This could support the closure of water sources on conservation and other rangeland properties that, while needing to be done carefully (Wallach & O’Neill 2009), might be effective in reducing the seasonal fluctuation of Emu densities. On the agricultural side, Western Australia’s eastern and southern Wheatbelt is becoming increasingly marginal, due to both climatic and economic changes (Van Gool & Vernon 2005), with escalating debt lowering farm viability (Wheatbelt NRM 2013). Some areas in the Wheatbelt are already being converted to other uses, particularly carbon sequestration (e.g. Carbon Conscious 2013).

Many of the agricultural land adjacent to the proposed Esperance extension was allocated to agriculture from the late 1960s to the early 1980s, in a poorly planned government programme that had minimal regard for agricultural viability or environmental impacts (Jasper 1984). That programme finally collapsed in 1983, leaving a legacy of marginal farms (Jasper 1984). Various soil and agricultural studies have subsequently documented serious concerns about the long-term viability of specific areas (e.g. Scholz & Smolinski 1996). Some property owners from allocations in the early 1980s, and adjoining the proposed fence extension, have already received ex gratia payments from the Western Australian Government because their soil types have been deemed unsuitable for agriculture or because government clearing controls aimed at reducing salinity were introduced over large areas of this recently allocated land.

Concluding comments

Barrier fencing is a management tool from an era where much wildlife was considered ‘vermin’ with bounties paid for their destruction. Not only is this an archaic concept, its effectiveness and economic benefits are questionable at best and counter-productive at worst.

The push for a total barrier fence around south-western Australia promotes a future where agriculture is somehow barricaded against the natural flows and rhythms of the Australian continent, with larger Australian wildlife categorised as ‘pest species’ and dealt with through industry-determined, taxpayer-funded ‘invasive species’ programmes. Such a retrograde approach to integrating agricultural production and biodiversity values clearly requires rethinking. These issues are problematic for the connectivity conservation programmes emerging across Australia, which generally strive to develop working landscapes where biodiversity conservation and commercial production can coexist. While much is already being achieved through cooperation, for those programmes to be successful, connectivity needs greater recognition in the decision-making of state and national governments on future land use and infrastructure.

In some areas, such as the margins of agriculture in Western Australia, achieving an equitable balance...
between the needs of agriculture and the needs of wildlife requires better integrated, more efficient and more humane techniques than government and industry currently propose. The challenge for government is to rise above sectoral lobbying and allocate public funds where they can most effectively achieve outcomes of long-term public good.

While we cannot fully resolve the vexed question of how wildlife persistence can be improved in agricultural areas, the available evidence suggests that barrier fencing is ineffective and has more negative environmental consequences than is generally appreciated. More open-dialogue and evidence-based approaches are needed if equitable and effective coexistence is to be achieved.

In relation to the existing barrier fence and proposals for its extension, considering the large investment of public funds required and the negative effects that such infrastructure is likely to have over a such a large area, a full environmental assessment by the Western Australian Environment Protection Authority, as well as a rigorous cost–benefit analysis that includes the environmental costs, are needed

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Frankham R., Bradshaw C. J. A. and Brook B. W. (2014) Genetics in conservation management: revised recommendations for the 50/50 rules, Red List criteria and population via-
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 covenants programs now seek to complement the comprehensiveness, adequacy and representativeness of the public reserve system, either stating 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
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).

<table>
<thead>
<tr>
<th>Standards</th>
<th>Description</th>
</tr>
</thead>
</table>
| Valuable        | • 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 | Public  
                  • must be statutorily defined and resourced  
                  Private  
                  • must be reserved in perpetuity  
                  • any change in management status must have Ministerial or statutory approval  
                  Indigenous  
                  • 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    | • 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 | • 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.

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

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).

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

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).

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

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).

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

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 covenancing 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 covenancing 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 the 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 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.
The covenant entered into must be perpetual.

The landholder must not receive money, property or any other material benefit for entering into the covenant.

The covenant must be entered into with a deductible gift recipient.

The market value of the land must decrease as a result of entering into the covenant.

The change in the market value of the land must be more than $5000 due to the covenant. If the decrease in value of the land is less than $5000, the owner will only be eligible for a deduction if the land was acquired not more than 12 months before entering into the covenant and had meet all the above conditions.

Essentially, the deduction is equal to the gap between market value after the covenant and that prior to the covenant; that is the decline in value due to the encumbrance on title. This change in value is determined by the Australian Government’s Valuer-General not by the actual market.

The Nature Conservancy (2008), in its submission to Australia’s Future Tax System Review made the following observations in relation to tax incentives for private land conservation at a national level:

“The tax treatment of gifts of property, and the establishment of conservation covenants was substantially improved in the last decade, with recognition of the value of the donation allowable as a tax deduction, apportionable over up to 5 years. However,
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 landowners 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 leveraged 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 covenantors are playing financially in protecting biodiversity and to legitimize this land use further, ensuring greater consistency in the rate relief offered to covenantors 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.

Acknowledgements

Thanks to Sue Stolton, Nigel Dudley and Kent Redford for originally commissioning and reviewing this paper, which informed a global report on private protected areas (Stolton et al. 2014). Thanks also to the covenanting agencies for providing data on their conservation covenants at short notice. Denis Saunders and Martin Taylor provided critical comments that greatly improved this paper.

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Private protected areas in Australia: current status and future directions


Payment for ecosystem services in practice – savanna burning and carbon abatement at Fish River, northern Australia

Nerissa Walton and James Fitzsimons

Introduction

Tropical savannas occur in Africa, Australia, South America and southeast and southern Asia. These ecosystems support about 10% of the world’s population, occupy one-sixth of the land surface and contain the most fire prone vegetation on earth (Russell-Smith et al., 2013). In Australia, savannas occur in the wet-dry tropics and feature both trees, dominated by eucalypts, and herbaceous plants, principally grasses. A defining feature of savannas is the existence of a dry season lasting up to nine months of the year. Grasses in the savanna have short, intense growing periods during the wet season, then cure rapidly during the dry. Rainfall is seasonal, ranging from 300 to 2000 mm with high inter-annual variation.

Indigenous people possess a long history of fire management prior to European settlement, living on their lands and maintaining skilled fire management over large parts of the landscape (Russell-Smith et al., 2009). This regime was disrupted with European arrival, which resulted in the movement of many Indigenous people away from their lands and the consequent breakdown of skilled fire management over large areas. Currently, over 20% of Australia’s northern savanna is owned or managed by Indigenous people and Indigenous land interests span a much greater area, through joint management arrangements and non-exclusive native title.

A feature of contemporary savanna fire regimes is the predominance of fires occurring in the late dry season, typically under severe fire weather conditions. Savanna wildfires can occur at any time in the dry season (March to November), however, 85% of fires occur in the late dry season, from August to November (Garnaut, 2011). Late dry season fires are characterised by low levels of patchiness, high intensity, high total fuel consumption and propensity to spread.

Contemporary northern Australian fire regimes are having significant impacts on regional biodiversity values (e.g. Woinarski et al., 2011) and contribute to national greenhouse gas emissions. Reinstating the traditional fire regimes for biodiversity, cultural and carbon benefits is now a rapidly increasing activity in northern Australia (e.g. Fitzsimons et al., 2012). Management of this significant ecosystem process presents great opportunity for ecosystem services. Here, we outline the Fish River Fire Project, the first early dry season savanna burning project to be declared under the Australian Government’s Carbon Farming Initiative and the first to have sold those credits, as a practical example of payment for ecosystem services.

The Fish River protected area

Fish River is a spectacular property with high conservation and cultural values in the Daly River region of the Northern Territory in Australia. It covers 182,500 hectares and encompasses sandstone ranges, large tracts of intact savanna, an extensive mosaic of monsoon forest wetlands, as well as the pristine waters of the Daly River and its tributaries. Fish River is home to unique assemblages of terrestrial and aquatic species and is an important refuge site for nationally-listed and Northern Territory-listed threatened species. It was purchased in 2010 by the Indigenous Land Corporation with support from the Australian Government’s National Reserve System Program, The Nature Conservancy, Pew Environment Group and Greening Australia (for more information, see Fitzsimons and Looker, 2012). As part of the National Reserve System it is managed primarily for biodiversity conservation and cultural values with other congruent objectives such as the management of visitor use, the needs of Indigenous people and local communities (including subsistence resource use) and to contribute to local industries such as tourism and the emerging carbon market. The tenure is a perpetual Crown lease for the purposes of ‘grazing and ancillary’ activities. The property is planned to be divested to Traditional Owners, upon the Northern Land Council’s resolution of the current native title claim.
Savanna burning on Fish River Station. Photo: © Ted Wood
The Carbon Farming Initiative

The Carbon Farming Initiative is a legislated offsets scheme of the Australian Government that allows farmers and land managers to earn carbon credits by storing carbon or reducing greenhouse gas emissions on the land. These credits can then be sold to people, business and government wishing to offset their emissions. Carbon credits are a financial commodity representing one tonne of CO₂ or carbon dioxide-equivalent (CO₂-e) that is sequestered and stored, or prevented from being released into the atmosphere, which can be sold as offsets. In Australia, the Carbon Credits (Carbon Farming Initiative) Act 2011 (Cth) and Regulations are the regulatory instruments that enable the production and trading of carbon credits and establish a set of integrity principles for Carbon Farming Initiative projects. These integrity principles require that abatement must be measureable and verifiable, additional to what would occur in the absence of a project, supported by peer-reviewed science and consistent with Australia’s international greenhouse gas accounts.

Within this framework, ‘methodology determinations’, the legal instruments that set out the rules for undertaking activities and measurement methods, are developed for each type of project activity, including for early dry season savanna burning.

Application of the savanna burning Carbon Farming Initiative methodology on Fish River

The Fish River Fire Project applies the Carbon Farming (Reduction of GHG Emissions through Early Dry Season Savanna Burning 1.1) Methodology Determination 2013. The methodology is based on traditional Indigenous knowledge coupled with western science which together demonstrate that controlled mosaic burning in the early dry season reduces fires in the late dry season and results in the avoidance of emissions of nitrous oxide and methane, both powerful greenhouse gases. Eligibility criteria for the methodology require the project to be on land receiving above 1000 mm average annual rainfall, containing the specified vegetation types, in this case eucalypt woodland and sandstone woodland, and having a history of regular late dry season fire.

Once eligibility is established, the legal right to carry out the project needs to be established and feasibility of the project assessed. The Northern Territory Government was approached to provide confirmation that the operation of an early dry season savanna burning project on the property was not inconsistent with the purpose of the Fish River lease. The Indigenous Land Corporation applied to the Clean Energy Regulator to become a recognised offsets entity and have the project declared an eligible offsets project. Once the project is approved, the activity, in this case early dry season burning, is undertaken. In some situations projects have been ‘backdated’ to include activity undertaken in the past and in the case of the Fish River Fire Project, project approval was backdated to 1 January 2011.

Under the savanna burning methodology, abatement is determined by calculating the annual emissions in the reporting period, and comparing this with the average annual emissions during the baseline period, the ten years prior to the project commencement. The abatement is the difference between the baseline and total emissions in the project year. Various tools are used to undertake the calculations, including a pre-validated vegetation map, seasonal fire maps from the North Australian Fire Information (NAFI) Service and the Savanna Burning Abatement Tool (SavBAT). Applications for Australian Carbon Credit Units are then submitted to the Clean Energy Regulator with an independently audited project offset report.

Impact of early dry season burning on greenhouse emissions

Prior to commencement of the project, 75% of Fish River burnt every year. Since the introduction of planned early dry season burning, the total area of the property burnt has reduced to 40%. Importantly, the proportion of the property burnt in the late dry season has reduced from 36% to 1% (Figure 1). This has resulted in significant avoidance of emissions of nitrous oxide and methane, which is converted to a carbon dioxide equivalent for carbon credit calculation purposes (Figure 2). A total of 49,041 credits have been issued to date, an average of 12,260 credits annually.

Selling credits on the market and media interest

The ILC has been issued with three tranches of credits to date, those for 2011 and 2012 combined and those for the 2013 and 2014 activity years. On each occasion, the ILC undertook a well publicised expression of interest process promoting the valuable cultural, social and biodiversity co-benefits associated with the project. Successful bids were identified based on three key criteria: the price; the organisation’s compatibility with Indigenous values; and how the company proposed to market the sale. Expressions of interest were received from liable entities (those corporations having an emissions liability under the Carbon Pricing Mechanism), carbon brokers and banks. Prices were high, recognising a genuine market value of the co-benefits produced by the project. The first two tranches of credits were sold to Caltex Australia for more than $22/t, a liable entity with a corporate social responsibility focus on Indigenous people and the environment. The media provided coverage to the expression of interest process and the sale of the credits. In particular, the ABC’s 7.30 NT program reported the first Indigenous owned Carbon Farming Initiative project and the first savanna burning project to sell credits on the market (Middleton, 2013).
Figure 1. Impact of early dry season burning on seasonality and area burnt at Fish River.

Figure 2. Impact of early dry season burning on greenhouse gas emissions – the red dashed line represents the baseline emissions for the 10 years prior to the purchase of Fish River and the green bars the emissions avoided from the baseline and therefore carbon credits produced. 2011 and 2012 abatement shown in this figure is calculated using the Carbon Farming (Reduction of GHG Emissions through Early Dry Season Savanna Burning 1.1) Methodology Determination 2013. Actual abatement claimed for these years under v1.0 was slightly less, as the baseline is lower when calculated under the first Determination. 2010, the year the ILC commenced management, is a transition year in the project, neither part of the baseline nor the project crediting period.
Land management costs

A deed with the Australian Government for Fish River requires that the reserve is managed in accordance with IUCN protected area category II (National Park). The types of costs incurred in managing the property include labour, infrastructure repairs and maintenance, weed management, fire management, visitor management, feral animal management, track maintenance and divestment consultations. Annual income under the Carbon Pricing Mechanism has represented approximately 1/3 of the annual cost of managing the reserve. Other complementary sources of income are the mustering of buffalo for pet meat, fire management on a neighbouring property, the ILC’s Real Jobs Program and contributions from The Nature Conservancy towards fire management and feral animal management.

Co-benefits of the Fish River Fire Project

The Fish River Fire Project is delivering social, cultural, biodiversity and economic benefits. All revenue from the sale of credits is reinvested in managing the property and supporting jobs and training for Traditional Owners. The employment of local Indigenous people, most of whom have familial connections to Fish River, is facilitating access for Traditional Owners to the property, reconnection with cultural values and protection of important cultural sites. The reduction in late dry season wildfire helps protect significant fire sensitive ecosystems and the many threatened species on the property, such as the Northern Quoll, Gouldian Finch and northern Masked Owl. The Fish River Fire Project is being used as a demonstration project to assist the development of other savanna burning projects by Indigenous groups in northern Australia.

Challenges and opportunities

Fish River was the first project to be accredited for the production of carbon credits from savanna burning and the first to sell its credits under the Carbon Farming Initiative. However, since registration of the Fish River Fire Project, a further 33 projects covering 13 million hectares have been approved by the Clean Energy Regulator and issued with over one million carbon credits as at January 2015 (Figure 3). This includes the registration of savanna burning programs that have been operating for a number of years (for example, EcoFire in the Kimberley, see Legge et al. 2011, and the West Arnhem Land Fire Abatement project in the Northern Territory).

The planned burning is undertaken about a year before credits are issued, so funds to undertake feasibility assessment, project development and burning are required prior to any income being received. This up-front cost could potentially inhibit more groups from entering the market without philanthropic or government support. Another challenge is land tenure, particularly the level of government involvement and consultation required to make land use decisions relating to Indigenous held land tenures.
The Clean Energy Act 2011 (Cth) was repealed in 17 July 2014 and the Carbon Farming Initiative Amendment Act 2014 (Cth), which gives legislative effect to the new Emissions Reduction Fund (ERF), commenced in November 2014. The Australian Government's ERF will buy the lowest cost emissions reductions offered through a reverse auction process, where bidders will be required to submit bids that offer up a specified quantity of emissions reductions from identified projects at a specified price. Although savanna burning is recognised under the ERF, a number of uncertainties are presented under this new system for new and current projects. Government auctions commence in April this year and it is hoped that a process valuing the important co-benefits achieved by these projects will be incorporated into the ERF in the future.

Conclusion

We consider the following key steps would contribute to furthering the policy direction for savanna burning in Australia:

- Recognition of Indigenous land managers as key custodians of land and providers of ecosystem services, including carbon projects, and consider support for capacity building and project startup on Indigenous-held land.
- A stable domestic carbon policy and compliance market, and/or a vibrant voluntary market to create demand.
- Society (community, private and public sectors) valuing the ecosystem services provided by nature. For example, valuing externalities, recognising the costs of unsustainable systems and product design in pricing. Appropriately valuing these services can create viable, sustainable opportunities for landholders to diversify their income, enabling “protection” and “production” to co-exist.

References


Links


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The Murray-Darling Basin Balanced Water Fund and the Environmental Water Trust – using markets and innovative financing to restore wetlands and floodplains in the Murray-Darling Basin for financial, social and environmental outcomes.

Ben Carr¹, Deborah Nias², James Fitzsimons³, Rich Gilmore⁴

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Key Points
- The Murray-Darling Basin Balanced Water Fund has been established and capitalized to achieve integrated financial, social, cultural and environmental outcomes.
- The Fund is the world’s first investor-funded solution to address water scarcity and account for the needs of irrigators, communities and nature.
- The Fund will donate through the Environmental Water Trust. Environmental watering projects will be planned and undertaken initially in the southern connected Murray-Darling Basin.

Keywords
Water fund, impact investing, environmental watering, water trading

Introduction
Water reform within the Murray-Darling Basin over the past 20 years has given rise to the opportunity for a private environmental water fund to be established using private investment to achieve multiple outcomes. In 2015, the Murray-Darling Basin Balanced Water Fund (‘the Fund’) was established with explicit financial, environmental, social and cultural objectives. The Fund is the world’s first investor-funded solution to address water scarcity and account for the needs of farmers, communities and nature.

The Murray-Darling Basin Balanced Water Fund
The Murray-Darling Basin Balanced Water Fund is the first water fund in Australia with the objectives of generating integrated financial, environmental, social and cultural returns on its investment. The Fund enables traditional capital market investors to support large-scale, long-term conservation works while diversifying their portfolio and earning income through investment in the Australian water market. The Fund was developed by The Nature Conservancy and is executed in partnership with Murray Darling Wetlands Working Group Ltd and Kilter Rural. Using investor funds, the Fund has acquired a portfolio of water rights through the existing Australian water market. Initial capital raising occurred in 2015 (see Kilter Rural et al. 2015) and the Fund as at May 2016 has AUD 25 million invested in approximately 8.5 gigalitres of water in the southern connected Murray-Darling Basin. Annual allocations from the Fund’s water entitlements are traded on a ‘counter-cyclical’ basis such that in the dry years when water is scarce and irrigation demand is high, more water is made available to irrigators. In the wet years when water is more abundant and agricultural demand is lower, larger volumes are made available to wetlands. This approach optimizes agricultural and environmental outcomes by replicating the natural wetting and drying cycles of the Basin. The fund uses the
Murray-Darling Basin Authority’s annual water determination to inform its annual watering strategy each year.

Environmental Water Donations

The water allocated to achieving the Fund’s environmental objectives will range from a minimum of 10% to a maximum of 40% of water owned by the Fund, and will be determined in May each year for the following irrigation year (1 July to 30 June) by reference to the volume of water held in major Southern Murray-Darling Basin water storages at 30 April (Table 1). On the basis of this criterion, the Trustee of the Fund will make a determination of its expectations for the forthcoming irrigation year, ranging from a water classification between Very Dry to Very Wet (Water Availability Classification). The determination of Water Availability Classification by the Trustee will determine the proportion of water assets owned by the Fund which it will donate for environmental purposes, to the Environmental Water Trust (see below).

Table 1. Donation % to Environmental Watering

<table>
<thead>
<tr>
<th>Water Availability Classification</th>
<th>Share Portfolio Assigned to Donations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wet/Very Wet</td>
<td>40%</td>
</tr>
<tr>
<td>Moderate</td>
<td>30%</td>
</tr>
<tr>
<td>Dry</td>
<td>20%</td>
</tr>
<tr>
<td>Very Dry</td>
<td>10%</td>
</tr>
</tbody>
</table>

The Water Availability Classification relies on historic data (the volume in storage at 30 April) to predict the level of water availability within the Southern Murray-Darling Basin in the coming irrigation year. To ensure that the amount of water donated by the Fund reflects actual conditions within the system, the decision rules contain an adjustment mechanism that enables the water donation in the following year to be adjusted by up to 10% where the actual level of water availability differs from that which was predicted. It is anticipated that over the Fund’s investment horizon, on average, approximately 20% of the Fund’s Water Entitlements will be donated for environmental watering purposes in the Southern Murray-Darling Basin wetlands and 80% will be traded through the water market in order to produce a return to its investors.

Environmental Water Trust

Depending on the annual water determination the Fund makes annual donations to the Public Fund of the Environmental Water Trust (EWT). The EWT is a tax deductible non-government environmental organization that was originally established by the Nature Conservation Council of NSW in 2007. Its objectives are to facilitate investment into the long-term environmental health of Australia’s rivers and wetlands. In late 2015, The Nature Conservancy and Murray Darling Wetlands Working Group Ltd (MDWWG) began a 10-year partnership through joint ownership and management of the Environmental Water Trust. Donations of water and/or funds to the EWT will be used to support an annual watering program developed by the MDWWG (see Nias et al. 2003).

Planning

Environmental watering will respond to seasonal conditions, water availability and ecological needs including targeting those wetlands, which do not receive environmental flows under current State or Commonwealth Government programs. A particular focus will be on floodplain wetland systems located on private lands which are essential for connectivity but rarely receive any water other than rainfall. The watering strategies will vary according to the level of water in the Southern Murray-Darling Basin, as detailed below.

Strategy 1 – Wet or Very Wet years: During a Wet to Very Wet year scenario, the entire environmental water allocation will be delivered into floodplains or very large wetlands. These environmental water donations will be added to natural flows to augment and extend the inundation of floodplain wetlands.
Strategy 2 – Moderate years: During a Moderate year scenario, water will be delivered into prioritized wetlands and creeks where appropriate infrastructure exists or can be developed efficiently. This strategy will target environmental watering of specific, smaller individual wetlands and creeks rather than broad floodplain inundation.

Strategy 3 – Dry or Very Dry years and drought: During a Dry Year scenario, small diversions of environmental water may be delivered into wetlands that provide critically important habitats, sustain areas of drought refuge for aquatic species, provide benefits of cultural significance, or improve water quality. In a Dry Year or Very Dry Year, environmental watering will be limited and in some years may not occur at all.

The Trust’s annual watering plan will be developed following determination of the water year and is aligned with a longer term five-year strategic watering plan. Strategic direction and advice is provided by a Scientific and Cultural Advisory Committee who will assist the EWT achieve targeted and strategic environmental and cultural outcomes. Watering objectives include Aboriginal social and cultural benefits and a range of conservation benefits are expected at both a landscape scale (flows to ‘harder to water’ wetlands on private land or floodplain forests) and local scale (improved health of key assets including vegetation, and threatened species), informed by identified flow requirements for species and ecosystems (e.g. Peake et al. 2011).

Complementarity to other environmental watering programs

The Trust provides opportunities for public/private sector partnerships and its environmental watering will complement government supported and run programs being undertaken by the Commonwealth Environmental Water Holder, Murray-Darling Basin Authority, state environmental water holders and regional natural resource management organisations. This complementarity will be achieved by 1) targeting high conservation value wetlands located on private land that are not currently targeted by existing government environmental watering plans, and/or 2) ‘piggybacking’ on environmental water provided by government organisations to enable extended watering of threatened floodplain systems that ordinarily would not receive water.

Projects

The initial project of the EWT was delivery of 950 megalitres of water to the Carrs, Cappits and Bunberoo Creek systems located 45 km west of Wentworth in the Murray Scroll Belt bioregion. This project was launched in April 2016 and involved the EWT providing capacity for the MDWWG to plan, administer and deliver water provided by the Commonwealth Environmental Water Holder. The watering is on land that is currently state forest that will be returned to Aboriginal ownership and management through the Tar-Ru Lands Board of Management. An eco-cultural monitoring program is currently underway to determine the outcome for both environmental and Aboriginal cultural values.

Conclusion

Delivering water purchased by government to stressed wetlands and rivers is an established key government strategy to address the decline of Australia’s aquatic ecosystems. The presence of arguably, the most mature and sophisticated water market in the world, combined with the significant knowledge and understanding provides opportunity for the private sector to become engaged in this strategy and contribute to the social, economic and environmental objectives of government and the community. This new model is relevant beyond Australia’s Murray-Darling Basin and were suitable conditions exist it could potentially be replicated in many water scarce geographies. The Nature Conservancy is currently in the process of scoping opportunities across the Western United States and Latin America where investor-funded solutions can address and balance the water needs of farmers, communities and nature.
**References**


Reforms required to the Australian tax system to improve biodiversity conservation on private land

Fiona Smith, Kate Smillie, James Fitzsimons, Bruce Lindsay, Gary Wells, Victoria Marles, Jane Hutchinson, Ben O’Hara, Tom Perrigo and Ian Atkinson

Private land conservation forms an integral part of Australia’s natural resource management and biodiversity conservation efforts, and the past two decades have seen a significant growth in the establishment of in-perpetuity conservation covenants. Specifically, conservation covenants address key national goals such as building the National Reserve System and expanding the markets for ecosystem services. However, a number of financial barriers exist to achieving these goals, and the national tax review in the form of the Tax White Paper Task Force provides an opportunity to address these barriers. This article provides a number of specific recommendations which outline how these financial barriers for private land conservation might be addressed by the Federal Government.

INTRODUCTION

In the last two decades, there has been increased emphasis on retaining and restoring ecosystem services on private land and creating innovative mechanisms and schemes to facilitate and achieve those outcomes. This has included mechanisms to create a market-based demand for ecosystem services on private land.¹

Private land conservation forms an integral part of Australia’s natural resource management and biodiversity conservation efforts. The efforts of private landholders are essential to meet the international Convention on Biological Diversity biodiversity targets, referred to as the “Aichi Biodiversity Targets”.² The role and importance of private landholders in biodiversity conservation is

² Convention on Biological Diversity, opened for signature 5 June 1992, 1760 UNTS 79 (entered into force 29 December 1993) <http://www.cbd.int/sp>. The 20 Aichi targets include: by 2020, at least halve and, where feasible, bring close to zero the rate of loss of natural habitats, including forests; by 2020, establish a conservation target of 17% of terrestrial and inland water areas and 10% of marine and coastal areas of ecologically representative, well-connected protected areas; by 2020, restore at least 15% of degraded areas through conservation and restoration activities.

(2016) 33 EPLJ 443 443
explicitly recognised in Australia’s Biodiversity Conservation Strategy, which seeks to increase the extent of private land managed for biodiversity conservation. The National Reserve System has been identified as the single most important asset for the conservation of Australia’s unique and globally significant biodiversity. The National Reserve System is Australia’s network of recognised protected areas. The goal of the National Reserve System is to develop a comprehensive, adequate and representative system of protected areas, to secure the long-term protection of Australia’s terrestrial biodiversity.

In 2011, non-government conservation organisations were identified as the fastest growing sector building the National Reserve System. In 2014, it was estimated that there are approximately 5,000 properties that can be considered private protected areas, covering 8.9 million hectares. Critically, these private protected areas conserve some of the nation’s most endangered ecosystems and species, and their protection via legislative and contractual mechanisms has saved Commonwealth and State governments considerable expenditure through not having to acquire this land themselves. For Australia to meet its obligations for creating a representative reserve system under the Convention on Biological Diversity, critical gaps in the reserve network will need to be filled. Most of these gaps occur in regions dominated by private land, where voluntary private land conservation mechanisms will be the only realistic options for filling the gaps.

Australia’s Biodiversity Conservation Strategy: identifies the importance of environmental markets and other incentives to achieve an increase in private landholders managing biodiversity and ecosystem services; recognises the need to encourage increasing private investment in biodiversity conservation so that both the costs and the benefits of biodiversity use are distributed across relevant sectors; and seeks to increase private expenditure on biodiversity conservation. As stated in the Strategy:

Society as a whole benefits, and future generations will also benefit, from protecting biodiversity. However these benefits are not fully reflected in our economic system. To ensure that biodiversity’s importance as a public good is fully valued, we need to ensure that there are financial incentives for actions that protect or enhance biodiversity and that the cost of damage to biodiversity is accounted for in economic planning. One way of moving towards such a system is to stimulate the development and expansion of markets for biodiversity and ecosystem services, including initiatives such as the Australian Government’s Environmental Stewardship Program, the Victorian Government’s BushTender program and the New South Wales Government’s BioBanking program.

The Biodiversity Conservation Strategy sets a goal of doubling the value of complementary markets for ecosystem services by 2015 and seeks to achieve the following outcomes:

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3 Natural Resources Management Ministerial Council (NRMMC), Australia’s Biodiversity Conservation Strategy 2010-2030 (Department of Sustainability, Environment, Water, Population and Communities, 2010) 44.
8 Fitzsimons, n 7, 19.
10 NRMMC, n 3, 3.
11 NRMMC, n 3, 42 and 44.
12 NRMMC, n 3, 41.
1. an increase in the use of markets and other incentives for managing biodiversity and ecosystem services;
2. an increase in private expenditure on biodiversity conservation; and
3. an increase in public–private partnerships for biodiversity conservation.  

A number of financial barriers exist to achieving these goals, and the national tax review in the form of the Tax White Paper Task Force provides an opportunity to address these barriers. These barriers have been identified in both past research\textsuperscript{14} and by the collective experience of member organisations of the Australian Land Conservation Alliance,\textsuperscript{15} who are at the forefront of permanent conservation on private land and whom the authors of this article represent. This article outlines how these financial barriers might be addressed by the Federal Government. The article does not address State jurisdiction land-based taxation issues (eg land tax impost or exemption) as they relate to conservation covenants, but simply notes the need for consistency and equity in the treatment of such landowners. Also note that such considerations may have relevance to any reform of Commonwealth–State taxation transfer arrangements, taxation roles and responsibilities arising from the work of the Task Force.

**Barriers for Private Land Conservation in the Income Tax System**

Private landowners who voluntarily establish land under conservation/sustainable management provide an important public service, often at considerable financial cost to themselves. The importance of this is emphasised by the fact that some of Australia’s highest priority conservation lands, eg coastal rainforests and inland grassy box woodlands, are now found mostly on private land.\textsuperscript{16}

Private conservation landholders are frequently unable to earn significant income from their properties (because it is protected for conservation and many future development rights are forfeited) but in most cases must still meet the costs of: rates and taxes; pest, weed and fire management; and fencing on land that is ultimately held for future and current social benefit.\textsuperscript{17}

Although governments recognise the importance of protecting land with high conservation value for future generations, and environmental and economic resilience, there is very little financial reward or public recognition for those landowners who choose to protect their properties and implement actions beyond their perceived or legislated level of duty of care.

Increasingly, the provision of economic incentives is integral to securing biodiversity conservation management, and financial assistance to private landholders through payments and concessions is recognised as an important motivator for private land biodiversity conservation.\textsuperscript{18}

\textsuperscript{13} NRMMC, n 3, 42.


\textsuperscript{15} The Nature Conservancy (Australia Program), Tasmanian Land Conservancy; Trust for Nature (Victoria); Nature Conservation Trust of NSW; Queensland Trust for Nature; National Trust Western Australia; and Nature Foundation South Australia.


There are a number of impediments to landholders who want to manage their land for conservation, and increasing private landholder participation in biodiversity conservation and environmental markets will require an improved recognition of the public interest character of ecosystem service payments.

Various but limited tax incentives currently exist for landowners who engage in private land conservation initiatives. These include income tax deductions and concessional capital gains tax treatment for entering into conservation covenants or other permanent protection instruments registered on title that are recognised by the Australian Tax Office (ATO), as well as deductions or concessions for landcare operations. While these measures are intended to provide tax benefits and incentives to land-based environmental activities, they do so in a very limited manner. Reform is required to address limitations and barriers to landholder contributions to maintaining and often restoring Australia’s biodiversity and natural assets on private land. Opportunities for reform are outlined below.

Limitations of covenant concessions

Deductibility of expenditures against taxable income where a taxpayer enters into a conservation covenant faces important constraints. First, it only applies where land value declines by more than $5,000, and this valuation must be independently undertaken under the authority of the ATO. If the loss of value is less than $5,000, the deductibility is only available if the land was purchased less than 12 months before the covenant was entered into. Second, deductibility is available where no consideration has been received for doing so. This is the case regardless of the amount (or proportion) of income or capital co-contribution the landowner may receive for entering into the covenant. Hence, where a landowner enters into a covenant as a component of participation in a market-based or incentive scheme, such as in the case of certain grants available under the National Landcare Program, or receipt of income under native vegetation offsets transactions, the income tax deduction is not available. Consideration in the form of a money incentive is received for doing so. Aside from issues of deductibility of costs against income for entering into conservation covenants, limited capital gains tax concessions are available where capital proceeds are received for entering into a covenant.

In comparison with international best practice, current Australian tax policy might be characterised as miserly in its failure to acknowledge and reward public interest dimensions to these transactions. It is reasonable to assume both private and public interest characteristics in covenanting transactions, whether or not money is received for them. Private benefits may be revenues or private access to environmentally valuable land. Public interest benefits include the protection, conservation

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22 Income Tax Assessment Act 1997 (Cth) subdiv 40-G.
23 Income Tax Assessment Act 1997 (Cth) ss 31.5(2)(c)-(d).
28 Income Tax Assessment Act 1997 (Cth) s 104.47.
29 In particular, a private interest is generated and recognisable in the transfer of funds from the funding body to the individual benefit of the landowner. There may also be recognised a private interest to the landowner in improving the environmental
and/or improvement of biodiversity generally, as well as provision of other ecosystem services, such as carbon sequestration or water quality. The public interest character of entering into permanent protection registered on title is only addressed under Australian tax law where covenanting occurs entirely as a gift.

The establishment of “split-receipting” for charitable “ecological gifts” in jurisdictions such as Canada has been one method of recognising and accommodating the public interest character of environmentally beneficial transactions. Under the split-receipting approach, a landowner can receive a payment or benefit for permanently protecting environmentally sensitive land, and at the same time receive a tax deduction spread over five years for any unremunerated value of the “land use and development rights” effectively given up (gifted) in establishing permanent protection. This approach recognises elements both of consideration and “gift” (donation) in conservation covenanting transactions, where the transaction includes disposition of a benefit to a registered charity below the fair market value of the land.

This tax law framework enables Canadian funding in support of on-title protection of high conservation value land to complement and work proactively with the tax law incentives for landowners donating ecologically sensitive lands to a qualified land trust. Liberal tax relief for donation of land under a conservation easement is also a well-established mechanism of environmental policy and revenue law under the United States federal income tax code. It has coincided with very significant growth in private land conservation through conservation easements and land trusts.

The policy basis of the Canadian approach is to allow for the private benefit of money or property received for entering into conservation protection to be distinguished from the “charitable” (public) benefit of encumbering the land for conservation purposes and reducing its fair market value.

The tax treatment in the Canadian approach better acknowledges and deals with the private/public benefit distinction actually operating in environmental incentive and market schemes. In Australia, such an approach would enable all jurisdictions to far more effectively leverage private contributions to the nation’s National Reserve System and natural capital bank. Further, the more mature tax rules and conservation practices in jurisdictions such as the United States can supply considerable experience and lessons in enactment of a more proactive conservation and restoration policy within the qualities of their land. But additionally, there is a benefit to the community as a whole in the environmental values of the land being well managed and/or restored (hence a public interest), including, eg, improved biodiversity outcomes, water quality outcomes, decreased erosion or salinisation problems, or greater capacity or efficiency in carbon sequestration on that land.

Reforms required to the Australian tax system to improve biodiversity conservation on private land

30 P Figgis et al (eds), Valuing Nature: Protected Areas and Ecosystem Services (Australian Committee for IUCN, 2015).


32 For a guide to the application of split receipting (and other tax implications) to conservation covenants, see A Hillyer and J Atkins, Giving it Away: Tax Implications of Gifts to Protect Private Land (West Coast Environmental Law Research Foundation, 2004).

33 Internal Revenue Code, 26 USC § 170(h).


35 That is, making a gift of the value of the land associated with foregone opportunities to sell it at the going market rate or developing it. Under the Canadian approach, the “gift” is the transfer of property and entering into a conservation easement (a conservation covenant) is considered to be a form of transfer of property. For further discussion regarding “fair market value”, see Trust for Nature, n 14, 63-64.
Australian income tax treatment of conservation covenants.\textsuperscript{36} Taking the lead from international practice, Australian tax law might take better account of maturing trends toward environmental markets, incentive payments, and private land conservation arrangements.

**Income tax treatment of expenditures for “landcare operations” and potential application to “nature conservation” activities**

Under Commonwealth income tax law, benefits only applying to primary producers are relaxed through deductions for “landcare operations”. Capital expenditure for landcare operations is deductible. This category of deduction is available to rural landowners using their land to carry on a business for a “taxable purpose”,\textsuperscript{37} as well as to those using the land for primary production. Taxable purpose includes for “the purpose of producing assessable income”,\textsuperscript{38} which, depending on a landowner’s individual financial and taxation circumstances, may include income from environmental market schemes and incentives. Whether activities funded under environmental market programs and agreements are “landcare operations” requires clarification.\textsuperscript{39}

Provisions for “landcare operations” capital deductions do not appropriately recognise the expenditures of landowners managing permanently protected private land for public benefit without any income or contributions towards the expenditures incurred in so doing. In addition, provisions for “landcare operations” predate widespread use of financial incentives and market mechanisms for delivery of ecological management and restoration of private land.\textsuperscript{40} It therefore appears timely and appropriate to revisit the structure and content of these capital allowances to adjust the meaning of “landcare operations”. Such adjustments could include:

- enabling landowners with conservation covenants to claim a “landcare operations” deduction from assessable income earned, regardless of source of income, and whether or not they are carrying on a business for a “taxable purpose” on the land;
- shifting the language of these provisions toward “ecological management and restoration” activities (or alternatively the “management and restoration of ecosystem goods and services”), as distinct from “landcare operations”; and
- establishing, as appropriate, the purposes of actions enumerated in s 40.635 of the Income Tax Assessment Act 1997 (Cth) as “ecological management and restoration” (or alternatively the “management and restoration of ecosystem goods and services”), as distinct from ameliorating land degradation.

**A stand-alone architecture for “ecosystem services” payments under income tax legislation**

Tax provisions relating to carbon offsetting measures are distinctly contained in a stand-alone architecture under Pt 3-50 of the Income Tax Assessment Act. This approach provides an element of coherence and relative simplicity to the tax treatment of carbon offsetting. There would be value in a similar stand-alone, coherent treatment of revenues from the management of “ecosystem services” as a category of economic activity. It is a category that might incorporate activities such as participation in conservation tenders, environmental offsets and grant programs with a view to creating a unified scheme for this sector of land management activities.


\textsuperscript{37}Income Tax Assessment Act 1997 (Cth) s 40.630(1)(b).

\textsuperscript{38}Income Tax Assessment Act 1997 (Cth) s 40.25(7)(a).

\textsuperscript{39}For discussion of ATO rulings regarding landcare operations, please refer to Trust for Nature, n 14, 61-62.

\textsuperscript{40}See New Business Tax System (Capital Allowances) Act 2001 (Cth) Sch 1.
OTHER FEDERAL TAX LAWS AFFECTING PRIVATE LAND CONSERVATION

Goods and Services Tax (GST)

At present, the sale of land used for business purposes, primary production or a residence is GST-free if the purchaser intends to continue to use it for those purposes, but the sale of land covered by a Conservation Agreement or Trust Agreement and used in perpetuity for private conservation is not.

Policy change to provide equality of treatment under the GST between owners of private conservation land and other landowners should be considered if the private conservation market is to become a major part of the solution to biodiversity loss.

Living bequest transactions and “discount sales” for philanthropic purposes

Covenanting organisations which are members of the Australian Land Conservation Alliance continue to be approached by individual landowners with high conservation value land wishing, for example, to continue to live on the property, but gift part or the entire property to a land trust for assistance with its ongoing management and future utilisation for land conservation purposes. In addition, landowners have also expressed interest in selling at a discount or “part gifts” of property to eligible community organisations. A “discount sale” is made when a donor sells property to an eligible organisation at a discount (ie below market value for a philanthropic purpose). Reforming the Australian income tax law to recognise “split receipting” for ecological gifting of conservation land will address the tax treatment vacuum surrounding living bequest transactions. Considering the reduced cost in purchasing this land (if deemed strategically important from a conservation perspective), this should be considered an important philanthropic contribution to society and this discount could be recognised as a gift for tax purposes. Currently, in the tax system, there is a lack of clarity around the more flexible approaches to environmental philanthropy which include living bequests and part-gifts or discount sales, thus providing a barrier to participation in these activities.

PROPOSED REFORMS OF AUSTRALIA’S TAX SYSTEM TO SUPPORT CONSERVATION OF BIODIVERSITY ON PRIVATE LAND

The issues in this article address the importance of increasing the number of landholders engaging in the sustainable management of private land in a way that protects biodiversity, provides ecosystem services, and manages private land for conservation and public benefits. Listed below is a series of proposed reforms to the federal tax system, which also have relevance for all Australian jurisdictions to ensure that Australia’s tax system supports the sustainable use of the nation’s natural capital as well as financial capital. The suggested reforms would also play a part in helping the Australian Government meet its international obligations to build the National Reserve System to achieve agreed levels of protection for Australia’s distinct biological communities and species, through enhancing conservation on private land.

1. Landowners who receive a payment or incentive for permanently protecting environmentally sensitive land should remain eligible to receive a tax deduction spread over five years for any unremunerated value of the “land use and development rights” effectively given up (gifted) in establishing permanent protection.

2. “Landcare operations” deductions under the Income Tax Assessment Act should be reviewed with a view to broadening the availability of concessions to include “ecological management and restoration” (or alternatively the “management and restoration of ecosystem goods and services”).

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42 Canadian (and US) approaches to the tax treatment of ecosystem services payments that seek to clarify and distinguish private and public interest dimensions to revenues, as well as allowing for ecological “gifting”, should be adopted as a means of improving Australia’s ecosystem service protection efforts, and enabling a greater level of transparency, fairness and landowner confidence in environmental contracts; see also Trust for Nature, n 14, 64.
3. In reviewing the deduction, non-capital expenditure should be included and all landholders with conservation covenants should be entitled to a deduction against assessable income for conservation works expenditure (capital and non-capital expenditure) whether funding has been received or not.

4. The Australian Government should exclude payments for conservation activities from taxable income where associated costs are not claimed.

5. The further professional development or guidance of tax advisers on the nature and implications of environmental market schemes should be supported.\[^{43}\]

6. A stand-alone treatment of revenues from the management of “ecosystem services” as a category of economic activity to support the overall governance of environmental markets and enhance public sector funding programs’ ability to leverage private investment in conservation should be considered.\[^{44}\]

7. Land protected by in-perpetuity conservation agreements registered on the title of land should be exempt from GST on future sale/purchase.

8. The circumstances where a landowner purchases land which is not protected by a perpetual conservation covenant with the intention of protecting its high conservation values, should be taken in account and legislation amended, ie A New Tax System (Goods and Services Tax) Act 1999 (Cth), to exempt land purchased for nature conservation from GST on the undertaking of the purchaser to place a perpetual conservation covenant within two years of the date of purchase.

9. Organisations constituted to establish in-perpetuity conservation agreements on property title, who purchase and sell properties for that purpose, should be GST-exempt.

10. The Australian Government should encourage “living bequests” of land with high conservation values able to meet the standards of the National Reserve System by clarifying that they are deductible (or rebatable) under the income tax gift provisions, and ensuring that any taxable capital gain at least excludes the value of retained rights or benefits.\[^{45}\]

**TAX AND THE FUTURE OF PRIVATE LAND CONSERVATION**

It has been identified that in-perpetuity conservation covenants on private land, including on farms, could become a significant contributor to meeting Australia’s national and international biodiversity obligations if provided with sufficient financial support.\[^{46}\] However, there exist significant barriers and disincentives for landowners in the current federal tax system for landowners considering placing permanent conservation covenants on their land and for those who already have land with such covenants. A tax system that removes financial barriers and provides support to private land conservation managers will both assist the Australian Government in fulfilling its conservation obligations in a cost-effective manner and recognise the significant role that private conservation landowners are playing in providing a public benefit. As stated by the Trust for Nature:\[^{47}\]

> Taxation law, policy and administration have an indirect regulatory role on PES [payments of ecosystem services] and environmental market schemes and private land conservation generally. For instance, not only may recipients of funds under environmental market schemes be liable to have those funds included in assessable income for Federal income tax purposes, but also the complexity and fragmentation of tax treatment of landowners participating in these schemes may have an impact on their capacity or desire to participate in the future.

\[^{43}\] ATO tax treatment information relevant to land managers undertaking landcare and conservation activities on private land could be simplified and made more informative and accessible to support landowners’ decision-making about participation in environmental incentive and market-based schemes: see Trust for Nature, n 14, 53.

\[^{44}\] Trust for Nature, n 14, 61.

\[^{45}\] Allen Consulting Group, n 41.

\[^{46}\] Taylor, Fitzsimons and Sattler, n 4.

\[^{47}\] Trust for Nature, n 14, 51. Refer also to landholder feedback regarding the complexity of tax implications for landholders.
More than 20 organisations use Conservation Action Planning (CAP), Healthy Country Planning and the Open Standards for the Practice of Conservation in over 140 projects, covering almost 160 million ha across Australia. This review documents the history, evolution and application of CAP in Australia and discusses its strengths, limitations and lessons learnt by users, including conservation planners, practitioners and policymakers.

Key words: Conservation Action Planning, conservation planning, Healthy Country Planning, participatory conservation, targets, threats.

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Introduction

For biodiversity conservation, the collective capacity to achieve outcomes increases with an adaptive management framework that informs and guides actions and measures and refines their effectiveness over time (Salafsky et al. 2002). Conservation Action Planning (CAP) is one such framework. Originally developed in the 1990s and introduced into Australia in 2001, it has now evolved and is used at multiple scales across public, private and Indigenous lands by a range of government agencies and not-for-profit organisations (Fig. 1).

In Australia, CAP and its adaptation designed for Indigenous conservation projects, Healthy Country Planning (HCP), are the tools and processes most commonly used to implement an approach now known internationally as the ‘Open Standards for the Practice of Conservation’ (herein ‘Open Standards’). The Open Standards are a globally recognised framework widely adopted across many countries and organisations which evolved from, among other things, early versions of CAP. For clarity, in this article, we use the term CAP broadly to describe the contemporary Open Standards approach, Healthy Country Planning and Conservation Action Planning.

While the adaptive management cycle is well known to most conservation practitioners, CAP uses the key steps of this cycle to develop a holistic programme that links desired outcomes to prioritised actions and resources, with appropriate measures of success (Fig. 2; Appendix S1). A number of tools and frameworks for conservation planning are used in Australia, including spatial prioritisation tools such as MARXAN and Zonation that seek to optimise the design of conservation reserve systems (e.g. Ball et al. 2009; Moilanen et al. 2009), and project evaluation and prioritisation (e.g. Pannell et al. 2012). These approaches tend to focus on either the initial strategic planning or the subsequent action that is necessary to achieve outcomes. CAP aims to do both and integrate these into a continuous review and improvement cycle. CAP therefore provides an overarching framework that can be used in combination with other conservation planning tools (particularly spatial prioritisation tools) as required.

A lack of literature on the Open Standards (and by inference, CAP) was identified by Schwartz et al. (2012). The purpose of this article is therefore to start filling these gaps by providing, from the viewpoint of our own long-term involvement in CAP and the Open Standards, a case study and literature review of its history, evolution and application in Australia. While a detailed evaluation of CAP in comparison with other conservation planning approaches or frameworks is beyond the scope of this article, we have sought to include here our perceptions of its strengths, limitations and some of the lessons learned by us and other conservation planners, practitioners and policymakers using CAP.

Purpose and History of Conservation Action Planning

What is Conservation Action Planning?

Conservation Action Planning is an adaptive management framework that guides the development of strategies, work plans and measures of success to achieve conservation impact. CAP is a practitioner developed and driven tool that seeks to balance speed, efficiency, accuracy and cost to provide a ‘credible first iteration’ plan of action to then be continually reviewed and adapted (Low 2003). Internationally, The Nature Conservancy (TNC) initially developed the approach in the early 1990s, largely in response to a broadening of its focus from site-based...

Since its development, CAP has been adopted and adapted by many organisations (Dudley et al. 2007). One of its historic adaptations was integration in 2002 with approaches used by WWF, Wildlife Conservation Society, Conservation International and Foundations of Success into what is now known as the Open Standards for the Practice of Conservation. The Open Standards are a framework to inform and improve conservation project design, prioritisation, management and monitoring to achieve success (CMP 2013). Their objective is to ‘bring together common concepts, approaches, and terminology in conservation project design, management and monitoring to help practitioners improve the practice of conservation’ (CMP 2015).

The Open Standards were released in 2004 by the Conservation Measures Partnership (CMP), a collaboration of conservation and funding organisations, that included WWF International, Conservation International, Jane Goodall Institute, Wildlife Conservation Society and The Nature Conservancy (for a full list of current members see http://www.conservationmeasures.org/about-cmp/members/). These organisations were seeking better ways to align the language, design, management and measurement of their conservation actions.

The approach has an explicit adaptive management structure that requires documentation of the assumptions on which decisions are based to support transparency (Schwartz et al. 2012). The Open Standards are common property, freely and openly available to conservation organisations worldwide (see http://cmp-openstandards.org/).

The CAP process involves identification of a project’s scope, its ‘targets’ (values), the systematic and strategic assessment of target viability, prioritised threats and strategies to address threats and/or increase the viability of targets. CAP asks three questions (Salafsky et al. 2002):

1. What should our goals be and how do we measure progress in reaching them?
2. How can we most effectively take action to achieve conservation?
3. How can we make conservation work more effective?

Conservation Action Planning is a process, specifically designed to be implemented and reviewed in an adaptively managed cycle of planning, doing and then reviewing (Fig. 2; further description of the Open Standards cycle that CAP uses is provided in Appendix S1).

The focus of the CAP framework when it was initially designed by The Nature Conservancy was on helping practitioners to decide what to do at a site or landscape scale rather than determining which site or landscape to work in. However, more recently in Australia, the framework is used, with the support of other tools, to both identify areas of interest within a specified scope and plan how these values can be conserved, providing a framework into which more detailed restoration and management planning can also be integrated.

Participatory processes in Conservation Action Planning

Conservation Action Planning uses participatory processes involving conservation planners, technical experts and scientists together with informed community representatives and practitioners with local expertise and knowledge (e.g. Dudley et al. 2007; Moorcroft et al. 2012). The process encourages the use of multidisciplinary and diverse teams to ensure that people with relevant skills and knowledge are involved at appropriate points in the process, for example ecologists in establishing indicators, strategic thinkers for strategy development and practitioners and managers for work planning and budgeting, respectively. The involvement of local people is often a key feature of the process and for Healthy Country Planning this involves local Indigenous communities and owners of land with traditional knowledge. The inclusive nature of CAP means that ‘the people who will ultimately be responsible for implementing the project must also be involved in designing and monitoring it’ (TNC 2007).

Typically, the initial phase of the process is a series of facilitated workshops led by a trained and skilled facilitator and involving experts, planners and practitioners at various times. Some organisations develop an initial values analysis to input into this workshop. The outputs of initial workshops are developed with contributions from specific subject matter experts, published reports, technical (including spatial) information/data and scientific knowledge incorporated to address knowledge gaps. The process uses results chains (program logic/theory of change) models to explicitly tease out how actions will lead to a desired result through a series of ‘if-then’ steps. Results chains involve a strategy, expected outcomes and the desired impact (Margoluis et al. 2013). The process and experience of thinking about how to achieve outcomes is valued as an important outcome in itself together with the final documented plan (Fig. 3).

History of Conservation Action Planning in Australia

Conservation Action Planning was introduced to Australia in 2001 by The Nature Conservancy, to help build the capacity and strengthen the strategic effectiveness of Australian conservation organisations. This was done through a series of training workshops and support to Australian organisations to apply CAP to the properties and landscapes in which they work. Initial training was designed to allow staff to share their experiences and approaches with others engaged in similar work. This exposure to the
process gradually generated interest and support from groups including Greening Australia, Bush Heritage Australia, Trust for Nature (Victoria) and programmes such as Gondwana Link.

By 2003, CAP was being used by the Trust for Nature (Victoria) and in 2004 work began on the first large-scale CAP for the Fitz-Stirling region of Gondwana Link (Box 1; Fig. 4), a Western Australian initiative by a consortium of restoration and conservation organisations. Greening Australia and Bush Heritage Australia, members of that consortium, also began using CAP for planning their conservation work in other parts of Australia.

By 2015, more than 20 organisations were implementing CAP in over 140 projects across Australia (Fig. 5). These range from the site/property scale through to landscape scale/cross tenure planning, public, private and Indigenous protected area planning and management, and species recovery (Table 1). Here, we outline the geographic scales at which CAP has been used in Australia by various organisations – along with its use for other planning objectives, such as Indigenous cultural values and individual species or species groups. The process of its delivery by different organisational types is also outlined.

**Different Applications of Conservation Action Planning in Australia**

**Geographically based application by a range of organisations**

**Site/property scale**

Conservation Action Planning is used as the basis of property management planning by an increasing number of conservation and restoration groups (e.g. Bush Heritage Australia, Greening Australia, Trust for Nature (Victoria), Tasmanian Land Conservancy and Nature Foundation SA) and government agencies (e.g. Parks Victoria). Bush Heritage Australia has adopted and institutionalised the CAP approach as the basis for their property and regional-scale planning framework (Walsh et al. 2013). They use it to determine what to protect, where, how and when to work, who will undertake the tasks, what resources and equipment they will need and how much time and money it will cost. Trust for Nature (Victoria) used CAP as the basis for management of their 30 000 ha Neds Corner property and surrounding public land in far north-west Victoria (Fig. 1) and for coastal lands around (Koch 2011) Port Phillip Bay and Western Port Bay.

**Catchment/subcatchment scale**

Conservation Action Planning has been applied at the catchment (watershed) and subcatchment scale to focus and prioritise conservation efforts. Examples include the Derwent Estuary Conservation Action Plan (which includes marine assets; Einoder et al. 2011) developed in 2012 by a group consisting of local governments, government agencies, universities, research groups and conservation groups. Some regional Natural Resource Management (NRM) groups (e.g. Natural Resources Northern and Yorke in South Australia, North Queensland Dry Tropics and Port Phillip and Westernport Catchment Management Authority) have adopted CAP as the basis of planning in their regions or subcatchment areas.

**Landscape scale**

At the landscape scale, CAP has been used for over 10 years as the basis for planning, managing and measuring conservation actions. In a review of large-scale connectivity projects in Australia, CAP was the most common framework used for planning across tenures (Fitzsimons et al. 2013). It was used to plan Australia’s major large-scale linkage programmes (some briefly described below) including Gondwana Link (Bradby 2013), Great Eastern Ranges Initiative (Dunn et al. 2012; Spooner et al. 2013), Habitat 141° (Carr 2013), Tasmanian Midlands (Males 2012; Cowell et al. 2013), Bunya Biolinks (Freudenberger et al. 2013) and South Australia’s NatureLinks (Gates & Kondylas 2013) (see also Walsh et al. 2013).
The flexibility of CAP provides an overarching framework for developing and delivering landscape-scale conservation outcomes (Beyer & Baker 2013). Some such initiatives used other conservation planning and analysis techniques in conjunction with CAP. For example, the Great Eastern Ranges initiative (Pulsford 2013). The plan identifies six targets:

- Creek systems
- Proteaceous-rich communities
- Tammar and Black-gloved wallabies
- Mallet and Moort woodlands
- Flat-topped Yate (or Swamp Yate) woodlands
- Freshwater systems

Threats to these targets that were identified included:

- Inappropriate fire management
- Predation by feral species
- Catchment clearing
- Invasive non-native alien species
- Fragmentation
- Pathogens including Phytophthora cinnamomi and other pathogens
- Cropping practices
- Grazing practices
- Development of roads or utilities

Four objectives for the Fitz-Stirling landscape:

- By 2012, restore at least 16 000 ha of native vegetation, including at least 2000 ha of proteaceous-rich communities that support native insect, bird and other vertebrate pollinators.
- By 2012, exclude stock grazing and manage foxes, other feral predators, plant pathogens (including Phytophthora cinnamomi), and invasive weeds over at least 60 000 ha of native vegetation in the Fitz-Stirling area.
- By 2012, significantly improve the condition of at least 60% of the creeks within the Corackerup catchment and, by 2017, within the Monjebup and Mid-Pallinup catchments.
- By 2017, increase the populations of Tammar and Black-gloved wallabies within the Fitz-Stirling area by 30%.

The Plan identified six strategic actions to achieve the agreed conservation objectives:

- Develop a landscape plan that identifies key areas for implementation of all strategies
- Purchase properties that most effectively deliver Gondwana Link’s ecological objectives
- Manage properties owned by Gondwana Link groups to demonstrate effective conservation practices in the Fitz-Stirling area
- Build integrated management across tenures through partnerships and other collaboration
- Restore native vegetation systems on geographically and ecologically suitable sites
- Reduce sediment and nutrient loads into creeks by rehabilitating erosion prone surfaces

Box 1. Targets, threats, objectives and strategic actions in the Gondwana Link Fitz-Stirling Conservation Action Plan (Gondwana Link Ltd 2008).
et al. 2013) used different methods (including CAP alongside satellite remote sensing and modelling to identify connectivity corridors, gaps and habitat fragmentation, ecosystem productivity, species richness, endemism and refugia) that varied according to planning scale, availability of relevant data and stakeholder interest.

Gondwana Link (Western Australia). One of the first large-scale applications of CAP in Australia was in the fragmented but largely agricultural areas of Gondwana Link in south-west Western Australia, between the Fitzgerald River National Park and the Stirling Range National Park (Bradby 2013; Bradby et al. 2016). Gondwana Link and its member organisations have consistently used CAP as the basis of their conservation outcome planning and implementation processes. Eight plans using the CAP framework have been developed for Gondwana Link’s focus areas, and a largely CAP-based Whole of Link Ecological Guide (Gondwana Link 2014) has been produced to provide an overview and context for the individual area CAP plans within the larger programme.

As an example of the approach, the Fitzgerald to Stirling Plan identifies six conservation targets, each with one or more ‘nested’, or sub-targets, nine threats, four objectives and six strategic actions (Gondwana Link Ltd 2008; Box 1). This CAP was instigated in 2004, has been reviewed and renewed a number of times since and is augmented by a spatial prioritisation (Lesslie 2012; Neville 2012) and individual site-based restoration plans (e.g. Jonson 2010). It continues to provide the collectively agreed basis for planning and conservation action in this landscape, is implemented by several organisations across the 240 000 ha area and is increasingly informed by focused research and an ecological monitoring programme that measures the response of the conservation targets to the ongoing management actions.

NatureLinks (South Australia). Conservation Action Planning was used in the South Australian NatureLinks initiative, with planning led by Greening Australia, including in the WildEyre conservation programme that is part of the East meets West NatureLink. The WildEyre CAP (WildEyre Working Group 2009) was initiated in 2008 by a range of groups who have been implementing the strategic actions identified in the CAP. The Living Flinders Conservation Action Plan covers the southern part of the Flinders Olary NatureLink with planning undertaken by a consortium of local groups led by the Greening Australia in partnership with the Northern and Yorke NRM (Berkinshaw & Durant 2012). Two other plans using the CAP framework cover the Yorke Peninsula and parts of the Northern and Yorke NRM. CAP has provided landscape level detail and interpretation of NatureLinks’ original high-level objectives and principles (Gates & Kondylas 2013). An important development arising from NatureLinks has been the integration of CAP into NRM planning by the Northern and Yorke Natural Resources Management Board (Northern and Yorke NRM 2014).

Application on Indigenous land: Healthy Country Planning

Healthy Country Planning (HCP) is the evolution and adaptation of CAP to improve its relevance and appropriate-ness for planning and management of country from an Indigenous perspective. The scope of HCP incorporates tangible and intangible values and ecological, cultural and socio-economic objectives (Davies et al. 2013). The HCP adaptations of CAP include the customisation of language to communicate the concepts, the inclusion of more socially and culturally relevant targets and the consideration of viability and threats from a cultural as well as an ecological perspective (Moorcroft 2012; Moorcroft et al. 2012) (Fig. 6). The first application of HCP arose through a partnership between Wunambal Gaambera Aboriginal Corporation, Bush Heritage Australia and the Kimberley Land

**Figure 4.** Looking along the Gondwana Link pathway in the Fitzgerald to Stirling planning area, showing some of the approaches being implemented following the Conservation Action Plan. Image: Green Skills and Airpix Photographs. [Colour figure can be viewed at wileyonlinelibrary.com]
Council to develop a plan for Wunambal Gaambera country in the Kimberley. The Wunambal Gaambera HCP guides management over approximately 2.5 million hectares of land and sea (Wunambal Gaambera Aboriginal Corporation 2010, reviewed in Austin et al. 2017).

Following this initial application, the Kimberley Land Council adopted HCP as their preferred planning approach to develop plans of management, which are required for the inclusion of Indigenous Protected Areas in the National Reserve System. HCP has been used in the development of eight indigenous land management plans within the Kimberley region up to mid-2016. Annual training workshops have been run in Northern Australia since 2011 to introduce Indigenous ranger teams and traditional owner groups to HCP, and train coaches to facilitate HCP. This has resulted in HCP being used by at least 30 indigenous groups across Northern Australia. More recently, there has been increased interest in HCP in central and southern Australia. In 2014, the Martu people completed a HCP covering 13.6 million hectares in the Great Sandy Desert, Little Sandy Desert and Gibson Deserts (Kanyirrinpa Jukurrpa

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**Table 1.** Summary of completed or initiated Conservation Action Plans (or similar) in Australia as on 1 December 2015

<table>
<thead>
<tr>
<th>Application of CAP by type</th>
<th>Number of projects</th>
<th>Area (million ha)†</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conservation Action Planning</td>
<td>111</td>
<td>90.35</td>
</tr>
<tr>
<td>Healthy Country Planning</td>
<td>31</td>
<td>65.07</td>
</tr>
<tr>
<td>Other Open Standards-based approaches</td>
<td>3</td>
<td>0.67</td>
</tr>
<tr>
<td>Totals</td>
<td>145</td>
<td>156.09</td>
</tr>
</tbody>
</table>

†Does not include projects where boundaries are unclear or unknown.
2014; Jupp et al. 2015), and in 2014, the Arabana people in South Australia completed a HCP over their country, including Kati Thanda–Lake Eyre. CAP has been highlighted as a recommended framework for IPA planning in the Australian Government’s Guidelines for Australian Indigenous Protected Area Management Plans (Hill et al. 2011). Across Australia, at least 25 HCPs are complete and another six are in preparation (Fig. 5; Table 1).

The degree of Indigenous community involvement in CAP, through HCP, provides a pathway to incorporate Indigenous language and core concepts, respecting and supporting community integrity, shaping ‘a more equitable intercultural conservation space’ (Moorcroft et al. 2012; Godden & Cowell 2016). A recent mid-term evaluation of the Wunambal Gaambera HCP (Austin et al. 2017) suggests that while broader community understanding of the HCP process can be limited, there are perceived benefits to the application to the resulting plans and processes. Beyond this review however, there has not yet been a detailed evaluation across all HCPs.

**Species-based application**

Conservation Action Plannings have been used for single species-based planning (e.g. South-eastern Red-tailed Black-Cockatoo) as well as for groups of threatened species. BirdLife Australia has recently used CAP as the basis for planning the recovery of a group of six threatened mallee birds in the Murray-Mallee region of south-eastern Australia. The Threatened Mallee Birds Conservation Action Plan recognised the efficiency of incorporating the same or similar actions identified within separate recovery plans and adopting an adaptive management framework to threatened bird recovery (Thomas et al. 2015). CAP has also been used to plan the recovery of a group of five species listed under the Environment Protection and Biodiversity Conservation Act 1999 found within the Mary River Catchment in south-east Queensland (Smith et al. 2012). The Mary River Threatened Species Recovery Plan recognised the value of the CAP approach in overcoming the ‘knowing-doing gap’ that occurs when existing knowledge about what needs to be done to achieve conservation is not translated into effective conservation action (Smith et al. 2012).

**How organisations are delivering Conservation Action Planning**

**Nongovernment**

Conservation Action Planning in Australia is supported and used by a range of organisations. The Nature Conservancy was instrumental in the growth and development of CAP through running a series of training workshops focused on training key personnel in the methodology and supporting these people to take the approach back to their respective organisations and fostering its uptake. The Nature Conservancy have run over 10 CAP training workshops across Australia since 2001, and these workshops have trained over 300 people across 30 organisations and have supported many groups and landscape-scale collaborations in using the methodology. The Nature Conservancy also have actively sponsored and supported the Healthy Country Planning (see above) adaptation of CAP and continues to foster Healthy Country Planning across northern and arid Australia.

Greening Australia worked with The Nature Conservancy to drive training and development of a network of users from 2006. Greening Australia uses CAP as the basis of its landscape-scale conservation programmes and other collaborative conservation projects involving diverse stakeholder groups. Projects using CAP have been running successfully for many years and have maintained strong and diverse partnerships that have survived changes in the political landscape and funding cycles. In addition to Greening Australia’s active contribution to Gondwana Link, examples of collaborative conservation programmes include the Pilbara region, Habitat 141°, Victorian...

**Figure 6.** Bunuba Rangers from Fitzroy Crossing in the Kimberley, Western Australia, learning the tools of Healthy Country Planning (Photograph: Natalie Davey). [Colour figure can be viewed at wileyonlineibrary.com]
Volcanic Plain, WildEyre, Living Flinders and Naturally Yorke.

Bush Heritage Australia uses CAP as the basis of their organisation planning and operations across their reserves and partnerships. They have integrated key elements of CAP including work planning and budgets, monitoring, analysing data, reporting and adapting into their business practice to improve their efficiency and effectiveness.

Government

Several government agencies in Victoria, Queensland, and the Commonwealth have promoted the use of CAP. Government agencies interest has developed through staff being exposed to CAP and through participation in CAP workshops. Wardrop and Zammit (2012) note that NGOs ‘testing and proving’ techniques such as CAP are often important precursors to government take up. Parks Victoria has adopted CAP following external audits (VAGO 2010, 2011), which highlighted the need to improve the links between expenditure on management actions and both the expected and achieved outcomes. The CAP process is progressively rolling out across 16 planning landscapes within Victoria, including the River Red Gum forests, Grampians, Wilsons Promontory and the Otways, and it has been used for most of Victoria’s 24 Marine National Parks and Marine Sanctuaries. Parks Victoria has found the CAP process particularly useful for its ‘asset-led’ approach and the development of logic chains between the goals for priority conservation assets, the management strategies to achieve those goals and the performance indicators required to measure change.

Parks Victoria’s experience to date is that the CAP process provides a strong step-by-step structure, and they are developing the most efficient and effective combination of desktop data analysis, expert consultation and use of structured workshops to engage with stakeholders in capturing knowledge and opinion and in establishing priorities (T. Varcoe, Parks Victoria, March 2017, personal communication). To improve guidance for investment, Parks Victoria has been exploring the use of other decision support tools such as Structured Decision Making, as an adjunct to the CAP process, particularly in the assessment of costs and benefits of implementing alternative strategies. Finally, Parks Victoria has also recently commenced a trial of the Miranda software (see below) for capturing and communicating the outputs of the CAP process at different geographic scales, and to evaluate its functionality and utility in embedding logic chain thinking in their work.

Our Reflections on the Strengths and Limitations of Conservation Action Planning

Below we outline the various strengths and limitations of CAP from our experience in Australia, informed by reports and published literature from within Australia and internationally.

Strengths

Focused on impact and explicit programme logic

Conservation Action Planning’s focus on the composition of the planning team, adaptively managing the process and the importance given to developing detailing work plans, budgets and monitoring all focus on achieving an impact. That is, CAP is ‘outcome-oriented’ rather than ‘output-oriented’. Consequently, the objectives developed during the strategy phase are always defined as:

- Specific – they target a specific area for specific levels of quantitative improvement
- Measurable – they quantify a readily measurable indicator of success
- Attainable – what realistically the project can achieve given the time and available resources
- Relevant – the objectives will most efficiently and effectively achieve the desired outcome
- Time-bound – specify when the result(s) will be achieved by.

Schwartz et al. (2012, p. 170) considered that one of the most compelling benefits of Open Standards is that it ‘require(s) practitioners to state specific goals that are measurable, impact oriented, realistic and time limited’. Recent analysis by Park et al. (2013) of the quality of targets developed by Natural Resource Management bodies in New South Wales and Victoria found the majority of their biodiversity objectives are not specific, measurable or time-bound. CAP provides tools and techniques that help address these shortcomings including a structured though adaptable planning framework, results chains and situation analysis to improve the realism of objectives and strategies and a recurring focus on defining objectives, implementation of actions, reviewing actions and then revising the plan in the light of the review.

The specific linking of ‘relationships among discrete actions, intermediate outcome and the desired final impact’ (Margoluis et al. 2013, p. 3) through using results chains is considered an integral strength of the CAP process. Results chains are a structured way of making cause and effect explicit and provide a basis for increasing understanding of why some conservation strategies will be more effective and efficient in achieving the stated objectives than others (Margoluis et al. 2013).

Participatory

Conservation Action Planning, as it is implemented in Australia, has a strong focus on inclusion that results in a process and set of strategies and actions that are recognised and owned by a diverse group of stakeholders. The benefits of this approach have been long recognised (Pressey & Bottrill 2009) and contrasts with ‘plan
Adaptable and flexible

Conservation Action Planning has been broadly adapted to apply to local circumstances. Schwartz et al. (2012, p. 172) states that ‘the Open Standards are compelling, in part, because they are general and flexible guidelines capable of being moulded to fit individual situations’. One of the most important adaptations in Australia has been the evolution of Healthy Country Planning. Guidance on incorporating climate change considerations into CAP has also been developed (TNC 2009; Game et al. 2010; Poiani et al. 2011). The Conservation Measures Partnership has developed adaptations to the Open Standards to incorporate human well-being and social targets (CMP 2012).

Conservation Action Planning’s use in Australia is independent of a legislative basis and the fact that is it not currently mandated through administrative practice has enabled it to retain a high degree of flexibility and ability to be customised or adapted by a particular user. Nonetheless, this would not exclude CAP’s use in a regulatory process.

A common language

Margoluis et al. (2013) considered that development of a common lexicon for the practice of conservation is a significant strength of the CAP process. The common steps and common language allow plans to be independently analysed, reviewed and combined or aggregated, facilitating peer review and candid exchange of the CAP process. This common language also facilitates communication about the practice of conservation and enables comparison and analysis of projects targets, threats, objectives and strategies.

Capacity and tools

The support of CAP through a worldwide Conservation Coaches Network is considered a major strength of the approach. The Network is made up of trained and experienced facilitators who teach the CAP methodology, guide participants through the process, facilitate and provide training in CAP workshops, and share lessons learned. Conservation Coaches provide the practical experience (including involvement in multiple CAP processes before becoming a coach) that is often necessary to achieve effective conservation results (Kareiva et al. 2014). The Network of Conservation Coaches is active in more than 60 countries across six continents and represents over 80 organisations. The Australian Conservation Coaches Network (CCNet Australia) of over 30 trained coaches (and many more in training) is a multipartner collaboration that supports developing and evolving the CAP methodology and ensuring that its use and application is tailored to Australian issues and conditions.

Specific software known as Miradi has been developed to support the CAP process and the management, storage and sharing of CAP information. Miradi is open-source conservation project support software designed by conservation practitioners, for conservation practitioners (Miradi 2014). Miradi is a tool for capturing, managing and sharing the outcomes from the CAP process. ‘Miradi Share’ is a web-based or ‘cloud’ database that allows exchange of conservation project information around the world and builds a network of people and a repository of knowledge focused on protecting the natural world (Bush Heritage Australia 2013).

Limitations

There are several recognised limitations in both the CAP structure and approach. Some of these are described as weaknesses; however, they often arise because CAP attempts to balance speed, efficiency, accuracy and cost to generate a ‘credible first iteration’ rather than comprehensive planning. The CAP process is not designed to provide highly specific, detailed analysis but rather generate adaptive management processes to inform and direct conservation actions in the most efficient manner, with frequent revision.

The Nature Conservancy (TNC 2011) have recognised some of the limitations inherent in CAP through an internal review in order to make it more relevant to emerging planning and conservation challenges such as the need to better integrate human well-being values. The following is drawn from that review and our own experience.

Insufficient scoping, planning and resourcing

The prerequisites for a successful CAP are (i) a shared common understanding of who the planning is for; (ii) who will ‘own’ the process and implement the plan; and (iii) a shared commitment to action and resourcing to implement the plan. The internal review found that sometimes these prerequisites are not sufficiently scoped, assessed, measured or resourced and this can contribute to failure or lack of engagement (TNC 2011). It is our view that this is more a limitation of the quality of the plan development, rather than a framework limitation. Solutions to this may include clearer guidelines to ensure those engaged in
commissioning and developing CAP are cognisant of the requirements and costs of undertaking CAP to the standard required for success.

Strategy selection

The process of strategy identification, development, testing, comparison and selection within the CAP framework has been identified as needing improvement (TNC 2011). This limitation was also identified by Low et al. (2010, p. 39) in a study of conservation planning framework used by public and private land managers in the western United States who found that ‘CAP lacks a methodology for actually optimising and quantitatively testing alternative strategies’. Both studies recognised the subjective nature of strategy and cost-benefit analysis used in CAP and the limited transparency in CAP’s decision support process. Game et al. (2013) suggested this is caused by the algorithms that are embedded in the assessment of risk and threats to targets in CAP not being explicit or easily modified within the tools that support the CAP process.

Wintle (2008), in a national review of biodiversity investment prioritisation tools used in Australia, noted that CAP lacks sufficient tools to enable comparison between alternative actions. Improvements that have been identified include more explicitly comparative frameworks that allow alternative strategies to be assessed and the most effective and efficient strategy that addresses a particular conservation need identified (TNC 2011). A specific weakness in the strategy selection process is the lack of specific Return on Investment criteria to allow value for money between alternative action strategies to be evaluated. Decision Analysis could also improve the process within CAP that support the selection of the optimum set of conservation actions (TNC 2011). To overcome this perceived weakness, Parks Victoria has trialled the use of Structured Decision Making as an additional decision support and optimisation step for strategy and action prioritisation (Walshe et al. 2013). In our view, while it is correct that the focus of CAP is on identifying strategic actions rather than an explicit process for determining priorities among potentially competing projects or areas, this can be resolved by integrating its use with other prioritisation techniques.

Insufficiently explicit spatial prioritisation

Some authors have suggested that the strategies and actions in CAP are not necessarily linked to spatially explicit priorities for investment in on-ground works (e.g. Wintle 2008). While CAP does not have its own explicit spatial tool, other spatial tools, including maps, are used extensively in the process of identifying targets, examining threats and developing strategies. We also believe the CAP process helps establish the key parameters for the development of spatial models – and reduces the likelihood the models are developed without a clear link to practitioner/stakeholder driven planning. In addition, there is no limit to the level of detailed planning that can be incorporated into or supplement CAP – a point that is addressed below.

Future Directions and Development of CAP in Australia

Spatial Conservation Action Planning

Since the development of the original CAP framework, there has been considerable development of spatial conservation planning tools. A number of organisations are examining how spatial information and tools can be integrated into the CAP framework including tools to analyse and combine existing spatial data to generate new spatial data that corresponds to elements of a Conservation Action Plan (TNC 2015).

A recent review of the Open Standards framework identified one of its key strengths was its ability to connect and interact with other planning approaches (Schwartz et al. 2012). For example, while the focus of CAP is on identifying strategic actions rather than an explicit process for determining priorities among potentially competing projects or areas, a wide array of complementary tools can be used in conjunction with CAP to make plans more spatially explicit and prioritised. For example, the Investment Framework for Environmental Resources (INFFER; http://www.inffer.com.au) is useful for prioritising among potentially competing on-ground natural resource management projects (Koch et al. 2010; Pannell et al. 2012). It was used successfully in conjunction with CAP in the Living Flinders project in South Australia (Greening Australia 2010) and as a tool for more fully developing, costing and prioritising projects initially identified using CAP. Schwartz et al. (2012) also recommended linking CAP and its support frameworks (e.g. Miradi) to existing spatial planning approaches (e.g. MARXAN) in a more explicit manner rather that adding spatial capacity to CAP or Miradi itself.

Accreditation

As the demand for CAP training (and coaches) increases, a formalised, industry-based accreditation process that supports development of CAP and establishes and maintains standards will be needed. Currently, CCNet Australia aids with training, mentoring and support to CAP coaches the global Conservation Coaches Network is in the process of developing a more formalised accreditation process for coaches than the informal certification currently in use. There is a role for universities to support formal accreditation through the Protected Area Learning and Research Collaboration (PALRC; http://www.palrc.com/) – CCNet coaches, as part of the PALRC
collaboration, facilitate training in the Open Standards as part of conservation planning and management courses at universities.

**Government uptake**

While CAP is being adopted by many nonprofit organisations and agencies, as outlined above, there has been increased interest from state, territory and Australian Government funding programmes as they have become more exposed to and see some of the uses and benefits of CAP in managing their delivery of conservation projects and funding programmes. There are opportunities for Australia’s 56 Regional NRM groups to make greater use of CAP in helping them plan and achieve both conservation and broader NRM outcomes in a structured, transparent and outcome-based manner.

**Conclusion**

Conservation Action Planning is increasingly being used at a range of scales for planning conservation actions in Australia. Its growth is due to many factors including the inclusive and participatory nature of the planning process and the adaptive nature of the planning cycle, and most importantly, its ability to focus on achieving the most effective conservation outcomes. The uptake of CAP, the diversity of users, locations and scales of application indicate that it is a useful tool that is able to be modified and supplemented with other processes and planning approaches.

The support provided through published and online resources and the integration of CAP in the global ‘Open Standards for the Practice of Conservation framework’ provides rigour, backup, resources and capacity to the CAP process. The availability of CAP coaches and an established network of Conservation Coaches (that itself is part of a global coach’s network that provide access to the knowledge and resources of a global team) offer significant assistance to users. CAP will continue to evolve, adapt and grow as a result of its use by a range of groups and organisations in Australia. The evolution of CAP to support the aspirations of Indigenous people in their land and sea management is a good example of this adaptation and evolution at work, and these developments are now being noted, promoted and exported worldwide.

**Acknowledgements**

We thank two referees for comments that improved this manuscript.

**References**


### Supporting Information

Additional Supporting Information may be found in the online version of this article:

**Appendix S1.** Components of the Open Standards cycle that CAP uses (source: Open Standards for the Practice of Conservation: http://cmp-openstandards.org/)
Trends and values of ‘Land for Wildlife’ programs for private land conservation

By Joshua A. Prado, Helena Puszka, Alexander Forman, Benjamin Cooke and James A. Fitzsimons

Introduction

Conservation of natural habitat on private land is seen as integral to maintaining Australia’s biodiversity (Fitzsimons & Wescott 2001; Robinson et al. 2011; Iftekhar et al. 2014); a viewpoint reflected in increasing global recognition of the importance of private land conservation (Norton 2000; Seigel & Lockwood 2010; Bingham et al. 2017). Approximately two-thirds of Australia’s land is managed by private landholders (ABS 2002), and for many threatened species and communities, core habitat occurs mostly on private land. Significant clearance of vegetation, particularly during early periods of European colonisation, has left landscapes fragmented, particularly in the southern areas of Australia (Bowers 1999). The conservation of biodiversity on private land is considered an important policy objective, and a number of programs have been developed to suit different circumstances (Figgis 2004; Fitzsimons 2015).

Land for Wildlife is one of several Australian private land conservation programs, including conservation covenanting and tender-based approaches, which aim to assist landholders in conserving and improving their property’s natural habitat. The program was established in Victoria in 1981 by the Victorian Government’s (then) Ministry of Conservation in partnership with the (then) Bird Observers Club of Australia and was substantially upgraded in 1990 (Platt & Ahern 1995a; Young et al. 1996). Land for Wildlife programs have since been taken up in most other Australian jurisdictions (Table 1). The program operates as a nonbinding voluntary scheme (McDonald 2001) broadly aiming to support landholders in providing habitat for wildlife on their property (DELWP 2015) (Box 1). Landholders owning property with pre-existing natural habitat may sign up for Land for Wildlife membership and receive certain benefits, including conservation advice; a Land for Wildlife sign to place on their property fence (Fig. 1); and access to educational workshops, newsletters and community events (Lines 2014; DELWP 2015). The program is inclusive, in that any farm, bush block, council reserve, school ground, golf course, cemetery and small or large property that can provide valuable habitat can be registered (Platt & Ahern 1995a,b). The program’s voluntary agreement allows for mixed conservation and production use of the land, and for participants to withdraw from the program at any time. The flexibility and non-binding nature of this program are considered to contribute to its appeal (Fitzsimons & Wescott 2001; Cooke 2012).

In the late 1990s, a number of reviews explored the potential for expanding the Victorian Land for Wildlife model to other Australian jurisdictions as part of a national template (ANZECC Working Group on Nature Conservation on Private Land 1997; Smith 1998) and an ‘Arrangement to coordinate Land for Wildlife Schemes’ was developed (Stephens 2000). The ‘arrangement’ was intended to facilitate a national approach to Land for Wildlife by setting out the principles...
and minimum standards that govern a Land for Wildlife scheme. The federally funded ‘Bush for Wildlife’ national coordination project was established in the early 2000s to facilitate a cooperative national approach to Land for Wildlife schemes and also to build on other programs and activities aimed at nature conservation on private rural land (Stephens 2000), although this role and function was short-lived.

Land for Wildlife programs are now established in nearly all Australian States and Territories, with South Australia and the Australian Capital Territory being the only exceptions (Table 1); a program has also been established in New Zealand. Delivery of the Land for Wildlife program functions differently in each State or Territory, as it is managed by a combination of government, nongovernment and private sector bodies (Table 1; Fitzsimons & Wescott 2001). Programs in other states operate on a memorandum of understanding with Victoria who owns the name and logo design. More formal arrangements to coordinate Land for Wildlife across the country have not been realised, and so informal coordination is currently undertaken (P. Johnson, pers. comm. 2017).

The Land for Wildlife model was the inspiration for a suburban equivalent, ‘Gardens for Wildlife’, which operates in some regions as a dedicated ‘sister program’ to Land for Wildlife (e.g. Tasmania, parts of NSW) or as stand-alone, nonaligned program (e.g. Albury-Wodonga, Knox City Council in Melbourne).

<table>
<thead>
<tr>
<th>State/Territory</th>
<th>Managing body(s)</th>
<th>Coverage</th>
<th>Delivery agent</th>
</tr>
</thead>
<tbody>
<tr>
<td>New South Wales</td>
<td>Community Environment Network (CEN)</td>
<td>State of NSW</td>
<td>Community Environment Network (CEN)</td>
</tr>
<tr>
<td>Northern Territory</td>
<td>Greening Australia</td>
<td>Top End</td>
<td>Greening Australia</td>
</tr>
<tr>
<td></td>
<td>Low Ecological Consultants</td>
<td>Alice Springs</td>
<td>Low Ecological Consultants</td>
</tr>
<tr>
<td>Queensland†</td>
<td>Natural Resource Management organisations, local governments</td>
<td>South-east Queensland</td>
<td>Brisbane City Council</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>City of Gold Coast</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Ipswich City Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Lockyer Valley Regional Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Logan City Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Moreton Bay Regional Council</td>
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<td></td>
<td></td>
<td></td>
<td>Noosa Shire Council</td>
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<td></td>
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<td></td>
<td>Redland City Council</td>
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<td></td>
<td></td>
<td></td>
<td>Scenic Rim Regional Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Somerset Regional Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>South Burnett Regional Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Sunshine Coast Council</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Toowoomba Regional Council</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Queensland Murray-Darling Committee</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Burnett Mary Regional Group</td>
</tr>
<tr>
<td>Tasmania</td>
<td>Department of Primary Industries, Parks, Water and Environment</td>
<td>State of Tasmania</td>
<td>Department of Primary Industries, Parks, Water and Environment</td>
</tr>
<tr>
<td>Western Australia</td>
<td>Department of Biodiversity, Conservation and Attractions</td>
<td>State of Western Australia</td>
<td>Department of Biodiversity, Conservation and Attractions</td>
</tr>
<tr>
<td>Australian Capital Territory§</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>South Australia§</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

†Various local governments formerly ran the Land for Wildlife program in north Queensland but no longer do. §No Land for Wildlife program is currently operating in this state or territory. A Land for Wildlife program on Kangaroo Island has been proposed.
Limited knowledge sharing and reporting on the operations of a program is a common situation for many of the voluntary private land conservation programs in Australia (Cooke 2012; Fitzsimons & Carr 2014; Fitzsimons 2015; Hardy et al. 2018a,b). Despite the popularity of the program indicated by the number of participating landholders and relative longevity of the programs, few reviews have been published since the national expansion of the Land for Wildlife program (for a review prior to the national expansion see Smith 1998). Nonetheless, a small number of studies on some aspects of the program have been conducted in some regions (e.g. Fitzsimons & Wescott 2001; Halliday et al. 2012). Here, we aim to provide the first national review of the Land for Wildlife programs, focussing on the growth, trends and value of the programs. This review focuses on potential to understand the aims, focus, strengths and challenges from the perspective of Land for Wildlife program coordinators in different jurisdictions, and how they might be improved to increase their effectiveness.

**Methods**

Quantitative data on the number of properties enrolled in Land for Wildlife, the size of properties and (where available) the area of retained habitat on those properties as at 30 June 2016 were obtained from each of the currently active programs. Other program details were obtained from telephone interviews with coordinators in all but one jurisdiction with Land for Wildlife programs. Not all programs distinguished between area of habitat retained and total property size. We also obtained information on the cumulative number of properties signed per year, from programs that had this information.

Structured interviews were also conducted with nine key representatives tasked with coordination or implementation of Land for Wildlife in the various jurisdictions. The interviewees selected were the most senior people coordinating Land for Wildlife within their jurisdiction/region, including representatives from three of the 13 local governments in South-east Queensland. This included three state coordinators (including the South-east Queensland Land for Wildlife coordinator), three local councils (three of which were in South-east Queensland) and three

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**Box 1. Common principles of Land for Wildlife programs**

(Adapted from DNRE 1997)

Land for Wildlife programs aspire to promote community participation in conservation and establish a conservation ethic among private landholders.

**Land for Wildlife programs provide**

- Free membership.
- Voluntary participation that is free of legally binding obligations.
- A kind of ‘club’ that offers information and ongoing support to landholders for management of habitat for native plants and animals.
- A diamond-shaped sign signifying program participation (Fig. 1).

**Land for Wildlife properties**

- Include intact vegetation or revegetated land, which may be being used for a variety of nonconservation purposes.
- Conserve species and habitats not adequately represented on public land.
- Provide continuity of habitat across landscapes.

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**Figure 1.** Land for Wildlife sign displayed at the front of a property in Victoria (Photograph: Ben Cooke). [Colour figure can be viewed at wileyonlinelibrary.com]
catchment coordinators. Representatives of NSW were not interviewed; nor were programs in Queensland beyond South-east Queensland.

The interview questionnaire involved 45 questions to identify coordinator views of strengths of their programs and areas for potential improvement (see Data S1). The interview questions were categorised on the basis of six indicators of the program’s growth and implementation: ‘success of the program’, ‘statistics’, ‘program details’, ‘landholders’, ‘biodiversity’ and ‘budget’. Open-ended questions were used strategically to enable the coordinators to elaborate on a topic (including strengths and challenges of the programs and whether national coordination would be beneficial), providing them with an opportunity to express their views on other aspects of the program they deemed significant. Interviews were conducted over the phone between 9 May 2016 and 20 September 2016 and lasted for approximately one hour. Answers remain anonymous to protect the interviewee’s privacy.

Once all the interviews were completed, we compared the responses of each coordinator within the table using the cross-tabulation method (Walliman 2006). This method allowed us to code the qualitative data used in this research. To prevent breaching the privacy agreements between the coordinators and the Land for Wildlife participants, no addresses were provided in the quantitative data provided with postcodes being the geospatial data that was provided.

Land for Wildlife operated in North Queensland in the late 1990s and 2000s, funded through the Natural Heritage Trust and managed by local government, then the Queensland Parks and Wildlife Service and Greening Australia (including the following local government areas: Townsville, Cairns, Cardwell, Cook, Douglas, Eacham, Herberton, Hinchinbrook, Johnstone, Mareeba, Thuringowa; Townsville City Council undated), but properties in North Queensland outside of these councils in North Queensland also have Land for Wildlife signs. These programs either no longer operate or their status cannot be definitively determined.

Results

Number and area

Overall, there are at least 14,000 properties registered as participating in Land for Wildlife programs nationwide (Table 2). There is a substantial difference in the number of properties registered in each program as at 30 June 2016, with Victoria having the most registered properties (totalling 5,537), and Alice Spring having the least (totalling 107 registered properties) (Table 2). Within South-east Queensland, which has 13 councils delivering Land for Wildlife programs, there is also marked difference in number between these local governments (Table 3). Land for Wildlife programs were established in different years and cover different extents of a state or territory (e.g. whole state, or particular region within state) (Table 2). Differences in number and area of Land for Wildlife properties in the different programs are likely to be a combined result of different years of establishment, resources available, underlying population density/degree of subdivision (and thus property size) and level of interest.

Table 2. Land for Wildlife programs, number of properties, total property area and retained habitat (where information available) and the year when program established

<table>
<thead>
<tr>
<th>State/Region</th>
<th>Number of properties</th>
<th>Total property area (ha)</th>
<th>Total retained habitat (ha)</th>
<th>Year established</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alice Springs (NT)</td>
<td>107</td>
<td>290,408</td>
<td>15,736</td>
<td>2002</td>
</tr>
<tr>
<td>New South Wales</td>
<td>1125</td>
<td>87,242</td>
<td>46,620</td>
<td>2001</td>
</tr>
<tr>
<td>South-east Queensland</td>
<td>3128</td>
<td>86,202</td>
<td>58,604</td>
<td>1998</td>
</tr>
<tr>
<td>Mackay (Qld)</td>
<td>186</td>
<td>21,970</td>
<td>NA</td>
<td>2000</td>
</tr>
<tr>
<td>Burnett Mary (Qld)</td>
<td>377</td>
<td>NA</td>
<td>NA</td>
<td>2009 (to 2013)</td>
</tr>
<tr>
<td>Queensland Murray-Darling (Qld)</td>
<td>349</td>
<td>46,051</td>
<td>NA</td>
<td>2000</td>
</tr>
<tr>
<td>Tasmania</td>
<td>996</td>
<td>59,054</td>
<td>NA</td>
<td>1998</td>
</tr>
<tr>
<td>Top End (NT)</td>
<td>207</td>
<td>18,787</td>
<td>NA</td>
<td>2010</td>
</tr>
<tr>
<td>Victoria</td>
<td>5637</td>
<td>526,862</td>
<td>140,624</td>
<td>1981</td>
</tr>
<tr>
<td>Western Australia</td>
<td>1941</td>
<td>1,195,607</td>
<td>284,541</td>
<td>1997</td>
</tr>
<tr>
<td>Total</td>
<td>14,043</td>
<td>2,332,183</td>
<td>546,125</td>
<td></td>
</tr>
</tbody>
</table>

†Only programs that recorded this separately are included. NA, data not available.

Table 3. Number of Land for Wildlife properties within South-east Queensland catchment municipalities (as at 30 June 2016)

<table>
<thead>
<tr>
<th>Municipality</th>
<th>Number of Land for Wildlife properties</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sunshine Coast Council</td>
<td>718</td>
</tr>
<tr>
<td>Brisbane City Council</td>
<td>423</td>
</tr>
<tr>
<td>Moreton Bay Regional Council</td>
<td>378</td>
</tr>
<tr>
<td>City of Gold Coast</td>
<td>345</td>
</tr>
<tr>
<td>Noosa Shire Council</td>
<td>282</td>
</tr>
<tr>
<td>Scenic Rim Regional Council</td>
<td>219</td>
</tr>
<tr>
<td>Logan City Council</td>
<td>183</td>
</tr>
<tr>
<td>Ipswich City Council</td>
<td>163</td>
</tr>
<tr>
<td>Lockyer Valley Regional Council</td>
<td>145</td>
</tr>
<tr>
<td>Somerset Regional Council</td>
<td>91</td>
</tr>
<tr>
<td>Toowoomba Regional Council</td>
<td>31</td>
</tr>
<tr>
<td>South Burnett Regional Council</td>
<td>6</td>
</tr>
</tbody>
</table>
Across the country, properties participating in Land for Wildlife comprise some 2.3 million ha (Table 2). Five programs (Alice Springs, NSW, South-east Queensland, Victoria and Western Australia) further distinguished ‘retained habitat’ from the total property area – totalling over 500,000 ha. Other programs either did not record ‘retained habitat’ separately from total property area or the information was not readily available at the time of data collection. Over half of this recorded ‘retained habitat’ is located in Western Australia (280,000 ha), despite the fact that state has <17% of the total number of properties from programs that recorded ‘retained habitat’ figures. This is reflected in the average size of retained habitat by jurisdiction (Fig. 2); regions with high proportions of rural land had significantly higher average sizes of retained habitat. Western Australia had the highest average habitat with 164 ha, followed by Alice Springs (147 ha) and New South Wales (41 ha). There is also a large difference in total area of registered properties in comparison with the area of the ‘retained habitat’ under Land for Wildlife agreements. SE Queensland had by far the highest proportion of conserved habitat relative to property size at 68%, 15% higher than the next program being New South Wales with 53%.

It is important to note the data from Alice Springs (Fig. 2) have been skewed to a higher level due to two significant parcels of land one measuring 200,000 ha and the other being around 20,000 ha. These combine to make up over three-quarters of the total property size of the Alice Springs program (see also Bridges 2012).

For South-east Queensland and Tasmanian Land for Wildlife programs, growth in the number of new properties signed per year has been steady; however, in Victoria there has been a recent flattening of growth of property registrations (Fig. 3). This slowing of growth could be linked to the reduction in resources that a number of states have experienced, particularly in regard to the number of staff employed in the program (see below). In the qualitative interviews, all but two of the participants listed their trend as ‘growing’. One described their program’s registrations as ‘currently stable’, but had growth in previous years when there were more resources, and another described theirs as ‘close to declining over the past 3 to 4 years’.

Annual budgets reported by the nine respondents varied greatly, ranging from $7,000 to $250,000 (the latter including wages), as well as two respondents who cited only ‘minor operational costs’ [Participant 7] or ‘no set budget … very minimal’ [Participant 2]. Four of the respondents considered that their budgets were stable, four stated this was unknown, and one cited a large reduction. While the majority of respondents (six) stated that their funding was ‘sufficient’, these respondents’ answers to other survey questions indicated that their funding was limiting, and two of these respondents added qualifiers: ‘[Funding is] currently [sufficient] but not enough to deal with a sustained increase’ [Participant 1]; ‘[There is] room for growth with increased resources’ [Participant 6]. Three respondents answered ‘No’ – funding was not sufficient. All participants described their
funding as secure or reasonably secure apart from one participant, who described it as ‘unsecure and hand-to-mouth’.

Staffing arrangements varied greatly across the nine programs whose coordinators we interviewed. Staffing was reported as ranging from as low as one 0.1 FTE staff member to 1 full-time staff member with five part-time staff members. No region had >1 full-time staff member. Three respondents indicated a reduced staffing trend for their programs, including reductions in full-time staff, part-time staff and a complete end to one volunteer coordinator program due to lack of funding. Overtime carried out by Land for Wildlife coordinators varied, from none (one respondent), almost never (two), occasional (four) and frequent (two). Only one respondent stated that their staffing resources had grown over time – others categorised their staffing resources as stable (three), reduction (one), large reduction (three) or unknown (one).

Extension and landholder engagement were the primary service offered by eight of the nine jurisdictions surveyed. Beyond site visits, workshops, information sheets and newsletters, some programs offered additional land management resources to their members, including seedlings, nest boxes, fencing and weed control. Three participants stated that Land for Wildlife officers in their program were also able to assist landholders in applying for grants offered by other organisations. One participant suggested their program, run through a local council, provided rate rebates to Land for Wildlife members. Almost all participants answered that landholders do not expect or request grants, and they understand the voluntary nature of the program.

We find that most people ... are more seeking the advice and technical assistance rather than financial support.[Participant 9]

Indeed, one program coordinator felt that providing grant funding might depart from the program’s focus on intrinsic stewardship.

I would see that [reimbursement for landholders] as a negative to the program, most likely encouraging the wrong people to the program.[Participant 4]

Concern that financial incentives could be a negative may reinforce the earlier point that Land for Wildlife’s central benefit lies in rewarding and recognising ongoing stewardship and intrinsic conservation motivations (although see Smith et al. 2016).

Landholder engagement

The two main methods of promoting the program were word of mouth and the Land for Wildlife signs on members’ properties, followed by field days and workshops, letter drops and flyers, presence at events, Land for Wildlife website and social media. Only two programs were proactive with reaching out to specific landholders: one reached out via phone, desktop audits and site visits, while the other sent letters to properties in high-biodiversity areas. No other participants conducted proactive Land for Wildlife promotion to specific landholders, but rather reacted to already interested landholders.

An initial site visit is usually the only site assessment that is completed on Land for Wildlife sites. Other site visits are conducted no more frequently than once a year. In some cases, site visits are only conducted if the landholders raise queries or concerns.

All participants stated that the Land for Wildlife sign, placed visibly at the entrance to most properties (Fig. 1), was significant, with participants naming it as ‘very pivotal’ [Participant 9] and ‘effective advertising and worn as a badge of honour by members’ [Participant 4]. It ‘allows landholders to advertise their conservation ethic’ [Participant 8] and is an effective form of promotion which raises questions from those who see the sign. Most programs indicated that, technically, a property is ‘deregistered’ upon sale of a property. However, in practice, there are few formal means of ‘deregistering’ a property if the landowner does not inform the program. This means that signs are likely to be present on some properties considered deregistered.

All participants reported conducting workshops and field days except for one, who had collaborated with other organisations to run them and experienced ‘a poor response’ [Participant 2] to these events. Another participant described their events as ‘quite popular’ [Participant 5] and that they generally book out for 20–30 people. Event frequency ranged from once a year to once a month, with the average frequency being four per year. Topics covered ranged from reptiles, weed management, providing habitat for frogs and snakes, wildflowers of the wet season, introduction to bird watching, junior rangers, feral animals, trapping, plant identification from botanists, nest box workshops, camera monitoring for fauna and landholder ‘walk and talks’ [Participant 9] lead by the landholders.

Relationship with conservation covenants

Most participants agreed that Land for Wildlife could be a gateway for landholders to conservation covenants (and many covenantors in different jurisdictions also have Land for Wildlife registration: Fig. 4). One participant suggested a strong gateway link between Land for Wildlife and covenants, with all of the 68 covenants in the area originally beginning as Land for Wildlife members. However, another participant stated that there did not seem to be much transfer from Land for Wildlife to conservation covenants in their region and noted the sensitivity with which this suggestion needed to be treated. This suggests strong regional differences regarding the extent of translation from Land for Wildlife to covenants:

Usually they’ve spent many years in Land for Wildlife, we’ve really engaged the landholders ... and they get to a point where they hate the thought that if they were to leave their property then something might happen to it. Then they start asking about permanent conservation agreements ...[Participant 6]
People have been in the Land for Wildlife program they tend to be quite happy there and haven’t thought to change their status . . . We let people know about it [covenants] but we certainly don’t push them, because we don’t want people to feel that the government is trying to take control of their land.[Participant 9]

This variation in landholder perception of conservation covenants in different regions may be attributed to different demographics, different attitudes towards government intervention or different government histories and reputation; although the small sample size and skewing of the data towards one region limit our potential to draw inference for all Land for Wildlife programs.

Targeted recruitment

Only one program stated that they target specific high-value habitat (Fig. 5). Three programs stated that they previously did this but have discontinued; one stated that they would like to, and one explicitly stated there were not enough resources to do so. Five of the participants had no strategy in place for targeting certain areas or sites, while the minority had corridor mapping and biodiversity strategies in place. Most participants stated they held no preference between high-biodiversity value and large property size.

If we were actively promoting [Land for Wildlife], we would probably go down that path, looking at biodiversity hotspots, trying to get groups of landholders together or corridors linking up, but we’re just not doing that at the moment.[Participant 5]

We sort of just sign up any property that has available habitat . . . We get them to sign up, support encourages them to revegetate the property.[Participant 7]

The limited capacity to target properties demonstrated in these responses shows a potential gap in the program’s effectiveness in regard to biodiversity outcomes.

Figure 4. Many Land for Wildlife-registered properties also have conservation covenants (from top: Victoria, NSW, Tasmania, north Queensland) (Photographs: James Fitzsimons). [Colour figure can be viewed at wileyonlinelibrary.com]
Value of national coordination

Opinions on whether there was value in more formalised national coordination of the program were divided. Four participants stated they saw value in national coordination and listed improved consistency, communication and coordination as benefits. Three participants did not see value in such an approach, listing the following reasons:

- Too challenging to be feasible.
- No clear advantage and no benefit to landholders.
- Large area to manage, a ‘big unwieldy bureaucratic problem’.
- It was very clearly articulated in the original Victorian Land for Wildlife agreement that it was not a national scheme.

These participants listed some alternate management changes, such as more coordination between branches, or an adoption of the South-east Queensland model to other states:

- The key is in the [South-east Queensland] model: putting facilitation through local government puts people in contact with the community. Land for Wildlife . . . at state level it becomes a bit big for facilitation outcomes on the ground level, whereas local governments have a pretty good beat on their local area, and can facilitate participation with landholders on those properties.[Participant 5]

In terms of program structure, South-east Queensland presents a different case to other regions. There is a coordinator in South-east Queensland that oversees the activities of 13 different local councils that make up that region. Regional coordination provides potentially greater efficiency through consolidating resources and thus sustaining the program which services all councils, including those that have as little as six registered properties, for which hiring their own Land for Wildlife coordinator would not be financially viable.

Strengths identified by Land for Wildlife coordinators

Over half of the survey participants (six of nine) listed membership from large properties as one of their regional program’s biggest successes, and four participants listed community education or increasing awareness of conservation as their biggest successes.

[Our biggest successes] are probably the fact that we are recognising continued contribution, [landholders] get some support, a sense of pride about conservation and rehabilitating their property.[Participant 7]

[Our Land for Wildlife officers’] level of expertise is incredible. They’re able to walk onto properties and identify pretty much every plant they see, and identify what the property management issues are. I think it’s a combination of their expertise and passion and ability to engage with landholders and get the trust of landholders.[Participant 6]

When [our region’s] program was restricted [with greatly reduced funding] in 2015, we had quite a large number of people write in complaining about that, even including questions in the houses of parliament. I think that reflects that there is quite strong support for Land for Wildlife and the principles that Land for Wildlife stood for . . . Had the same sort of thing happened 20 years ago you wouldn’t have heard a murmur. I think that’s demonstration of the success of the program in changing attitudes of the community.[Participant 9]

These responses suggest Land for Wildlife coordinators see a range of benefits associated with the program. These responses also indicate that the Land for Wildlife program has value beyond purely conserving native habitat.

When we have not surveyed Land for Wildlife landholders themselves, the sentiments above suggest coordinators feel the programs may be acting to change community attitudes to conservation, build landholder knowledge or reward long-term commitment to conservation on the part of the community.

Over half of participants (seven of nine) believe that the extension support that Land for Wildlife brings to landholders is one of the program’s key points of difference from other similar programs in
their region. This support includes propri-
erty-specific advice and support material,
personal one-on-one contact and a
‘broad’ (Participant 2), ‘comprehensive’
(Participant 1) and ‘holistic’ (Participant 8)
approach to land management, with a
focus beyond just bushfire preparation,
weed management or other areas such as
erosion and revegetation.

Challenges and suggestions
identified by Land for Wildlife
coordinators

Despite six of the nine respondents sug-
gestng there was sufficient funding to
maintain their program at its current level,
the most common challenge to program
implementation (also represented by six
participants) was insufficient resources
(which included limited staff numbers,
time constraints, budgets and limited ability
for site visits). Maintaining the database or
keeping on top of members who move
address was mentioned by two participants
as challenges. To address resourcing issues,
two participants suggested a partnership
model of delivery with other conservation
groups, including partnerships with NRM
regions. Two other participants suggested
a more centralised approach, one of these
suggesting a review of national guidelines,
more national consultation and policy, and
centralised data.

Other suggestions, each mentioned
once, included:

- The ability to access experience and
  knowledge from other branches.

- Use of orange signs to indicate proper-
ties that are ‘working towards’ mem-
bership (a feature of the program in
New Zealand).

- A lower grade of membership available
to properties, such as suburban lots
with remnant vegetation on very small
properties that are not currently the
focus of the program.

- Making Land for Wildlife a stepping
stone towards binding conservation
covenants (while others expressed
reservations on this based on coordina-
tor observations of some landholders
being dissuaded to signing Land for
Wildlife if a link to permanent agree-
ments is seen).

- A central website with links to all regi-
nonal Land for Wildlife websites and
resources.

- Yearly reviews of Land for Wildlife, pos-
sibly including a face-to-face confer-
ence with visits to Land for Wildlife
properties.

Discussion

Land for Wildlife programs are bound by
common objectives of encouraging landowners with wildlife habitat on their
properties to manage it for conservation
(Box 1), with loose national coordination
of the programs. Nonetheless, governance
and delivery structures vary between
regions, as does the number, area and
average sizes of properties. While there
were many commonalities in responses
from coordinators, differences in
approaches were evident from respon-
dents, in part influenced by resources (fi-
nances, staff), geographic scope and
different histories of establishment.

While the sample size and higher represen-
tation by one region (South-east
Queensland) potentially limit the interpre-
tation of the questionnaire data with
respect to all programs, the results repre-
sent a range of conditions and views that
provide valuable insights, particularly
when combined with the quantitative data
which were gained from all but one
region. This allows us to draw on the data
and apply our own experience of private
land conservation programs to propose
future directions for the Land for Wildlife
program and research needs.

Future directions for Land for Wildlife

The expansion of Land for Wildlife over-
seas (i.e. New Zealand) and the extension
of the model into urban areas (Gardens for
Wildlife schemes, e.g. Mumaw 2017;
Mumaw & Bekessy 2017) highlight the
continued interest in the Land for Wildlife
model. Our results indicate the need to
bolster the alignment of the dispersed pro-
grams with the original vision and
agreement, particularly in some states
but also some regions where participation
is low. This is particularly vital given that
Land for Wildlife represents the preemi-
nent entry-level program for biodiversity
conservation on private land in Australia.

Based on the results and other observa-
tions, we recommend program reinvigora-
tion could be performed in a number of
ways:

- A more formalised national model of
  coordination that includes greater and
  more consistent resourcing that would
  enable the program to tap into state
  and federal funding and policy initia-
tives. This should not come at the
  expense of local/grassroots emphasis
  and identity of the scheme, but could
  enable a more targeted approach to sign-
ing up properties with regional, state or
  nationally significant biodiversity val-
ues. A particular benefit of greater
  national coordination would be the
  capacity to fund stewardship activities,
given their central role in the delivery
of Land for Wildlife. More formalised
  national coordination has been in place
for Landcare (through the National Land-
care Network) for a number of years and
for covenanting land trusts (through the
Australian Land Conservation Alliance)
in more recent times.

- In a number of jurisdictions, there are
  active networks of Land for Wildlife
landholders on social media (i.e. Face-
book (https://www.facebook.com/LfWAl-
cie/)) sharing information on the
wildlife and natural values of their
land.

- There is a need to re-engage with land-
holders who still have Land for Wildlife
signs but where the landholder is no
longer active in undertaking conserva-
tion; the property itself is no longer reg-
istered due to property turnover, or the
program has ceased to exist, for exam-
ple (e.g. parts of north Queensland;
Fig. 6). Where signs are present on
unregistered or orphaned properties,
there is a clear opportunity for initiating a relationship with the current landholder or reinstating a program in that region.

Further research

As the first analysis of Land for Wildlife since it expanded beyond Victoria, the findings raise a number of further research questions that are important to explore. Surveys of landholders in different programs, representing a wide geographic spread, should be undertaken to gain viewpoints from Land for Wildlife landholders themselves. This would be especially valuable to understand landholder motivations for registering for the program and experiences since then [as has been performed for landholders with conservation covenants (Stephens et al. 2002; Kabii & Horwitz 2006), fixed-term agreements (Burmeister et al. 2006; Cooke & Corbo-Perrins 2018) or multitenure conservation initiatives (Fitzsimons & Wescott 2007)]. Determining the extent to which Land for Wildlife acts as a stepping stone to more permanent conservation agreements is an important research question. Evidence of this is patchy and views from Land for Wildlife coordinators mixed, and more work needs to be performed to understand whether, and in what circumstances, Land for Wildlife paves the way for other conservation agreements.

The spikes in registrations among programs also need to be further explored to understand the situations that have historically led to increases and decreases in registration numbers in particular years. Exploring reasons why some programs thrive and others may not continue is also an important area of future research.

Finally, analysis of comprehensive GIS data on Land for Wildlife properties will allow for an assessment of the contribution such properties make to the management of particular ecological communities and threatened species, to better understand the conservation impact of the programs.

Conclusion

While there has been an increased focus on the growth of permanently protected private land (conservation covenants: Fitzsimons 2015; Bingham et al. 2017) and tender-based fixed-term agreements that target specific ecological values (Rolfe et al. 2017), the majority of landholders with habitat values on their land would not necessarily qualify for these schemes. Land for Wildlife programs provide the ability for landholders with a range of habitat values on their land to be recognised for the contribution they (and their properties) are making to conservation. With more than 14,000 properties registered with Land for Wildlife programs, this would make it the second most subscribed private land conservation program (behind Landcare) in Australia. In some programs, a decline in resourcing has seen diminished activity. We believe there is significant scope to revitalise Land for Wildlife in jurisdictions where it has been stagnant, to improve national coordination and profile and to enable better accounting for important biodiversity assets being managed for conservation on private land.

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Monitoring and evaluating the social and psychological dimensions that contribute to privately protected area program effectiveness

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ABSTRACT

Privately protected areas (PPAs) make important contributions towards global conservation goals. As with any protected area, PPAs must be monitored for effectiveness at protecting and managing biodiversity. However, the key drivers of maintaining and improving the effectiveness of PPAs are often social, particularly for conservation covenants and easements that are owned and managed by private landholders. In Australia, we surveyed 527 covenant landholders across three states (New South Wales, Tasmania, and Victoria), to provide a benchmark for monitoring and evaluation activities. We found that landholders are mainly motivated to participate in order to protect their land in perpetuity, but come to expect financial and technical assistance as a benefit of the program. While 71.1% (n = 344) reported achieving their land management goals, 44.7% (n = 242) of landholders struggle with covenant management because of age, and financial and time constraints. Covenant landholders are generally satisfied with the program (92%). A subset (8%) of landholders feels disaffected with their participation, relating to their perceived inability to personally manage the biodiversity on their land, and the lack of interaction they have with representatives of covenanting organizations. Where compliance monitoring and semi-annual technical assistance is limited, some landholders are concerned that the efficacy of the covenant is reduced. To increase effectiveness we suggest that PPA programs regularly monitor landholder satisfaction and management needs, schedule conservation actions based on landholder capacity, and utilize landholder networks to spread information and foster communities of stewardship. Additionally, given the older demographics of landholders, programs should engage in PPA successional planning.

1. Introduction

Privately protected areas (PPAs) are recognized as an integral, though perhaps under-appreciated, approach to conservation and sustainable development (Bingham et al., 2017; Stolton et al., 2014). Expanding and managing the protected area (PA) estate onto privately owned land presents challenges different to those faced by government-owned PA networks. PPAs offer opportunities for conservation organizations to work collaboratively with landholders and potentially improve cost-efficiency of conservation funding (Main et al., 1999). However, the contexts in which PPA programs operate are complex, dynamic and diverse (Cooke et al., 2012), and to an extent, the decision-making and behavior of landholders can determine the degree to which ecological values are enhanced or compromised. As such, maintaining the social and ecological values embodied in PPAs necessitates ongoing monitoring, evaluation and refinement of these initiatives.

The most common approaches to the monitoring and evaluation of
PPAs have focused on those that are government operated, e.g., impact evaluations (Ferraro and Pattanayak, 2006; Hockings et al., 2006; Stem et al., 2005). These approaches have the benefit of providing an audit of the ecological values within the protected area (i.e., species, ecosystems, ecological and evolutionary processes, and/or ecosystem services), and ideally the management processes designed to maintain them (Cook et al., 2010; Hockings, 2003; Knight et al., 2011), both essential components of evaluating PPAs. However, for PPAs it is often the decision-making and behavior of landholders that typically determines the degree to which the ecological values are retained or improved. As such, understanding, fostering, monitoring and evaluating a landholder's commitment and capacity to manage their land as well as the land they manage, is fundamental to establishing and maintaining effective PPAs.

Previous studies have assessed the social dimensions of PPAs or other in perpetuity conservation mechanisms for privately held lands. A large amount of research has been devoted to understanding landholder motivations for adopting PPAs (e.g., Ernst and Wallace, 2008, Farmer et al., 2011, Horton et al., 2017, Kabii and Horwitz, 2006, Pasquini et al., 2010). Smaller bodies of research have examined landholder satisfaction with specific PPA programs (Feinberg, 1997; Forshay et al., 2005; Selinske et al., 2015; Stroman and Kreuter, 2016), management activities and post-enrollment needs (Farmer et al., 2017; Fitzsimons and Wescott, 2007; Stephens et al., 2002; Stroman and Kreuter, 2015) and the challenges landholders face in managing their lands to achieve conservation goals (Fitzsimons and Carr, 2014; Halliday et al., 2012; Horton et al., 2017). What is missing is a holistic representation of how these social dimensions interact with PPA programs to maintain landholder PPA stewardship.

1.1. The dimensions of PPA stewardship

In addition to its conceptualization as an ethic or moral philosophy (Leopold, 1949; Enqvist et al., 2018), stewardship has been defined as ‘actions taken by individuals, groups, or networks of actors, with various motivations and levels of capacity to protect, care for or responsibly use the environment in pursuit of environmental and/or social outcomes in diverse social-ecological contexts’ (Bennett et al., 2018: 599). Within a PPA context, both the landholders and PPA organizations are actors working collaboratively towards the stewardship of private lands (Cooke et al., 2012), though much of the management responsibility falls on the landholder. Designing and delivering effective PPA programs requires organizations to motivate private landholders to enroll their land, while also continuing to facilitate and monitor landholder's PPA stewardship post-enrollment (Selinske et al., 2017).

Previous research has found the capacity of PPA landholders for undertaking stewardship (time, money, physical abilities and knowledge) is heterogeneous (Kabii and Horwitz, 2006). Landholder motivations for stewardship can be both intrinsic (e.g. sense of purpose, conservation identity) and extrinsic (e.g. access to management expertise, financial opportunities, recognition) (Pasquini et al., 2010). Multiple mechanisms (e.g. tax incentives, stewardship officer outreach, protection in perpetuity) can be used to cater for the variety of motivations among a population and within individuals, and to build landholder capacity for PPA management (Selinske et al., 2017; Young et al., 1996). How well landholder expectations of program performance are met (e.g. regular stewardship officer visits, timely and efficient establishment of the PPA) shapes their experience in participating, as do the benefits (e.g. psychological wellbeing, financial, expansion of social networks) they perceive to be gaining (Cross et al., 2011; Farrier, 1995, Horton et al., 2017 Selinske et al., 2015). Over time, these factors may impact a landholder’s satisfaction and so determine their commitment to the program, (as per conservation volunteer programs; Asah and Blahna, 2013; Miles et al., 1998). However, the factors that drive satisfaction may differ from those that motivated them to join the PPA program (Horton et al., 2017; Selinske et al., 2015), and are likely dynamic as a result of changes in the program, its context, and landholders needs (Lindsay, 2016).

We conceptualize PPA stewardship as long-term retention of a landholder in a program who exhibits compliance with a management agreement and to the agreed land use restrictions of the covenant. Although landholders managing covenanted PPAs may have in perpetuity agreements with a covenant/easement organization, there is no guarantee that they, or future landholders, will comply with their management agreements (Fitzsimons and Carr, 2014, although see Hardy et al., 2017). Buyers and inheritors have been known to be less enthusiastic about meeting management agreements and might be more likely to breach an in perpetuity contract (Collins, 2000; Rissman and Butsic, 2011). This presents compliance and enforcement issues for administering organizations, as well as questions for ecological integrity (Schuster and Arcese, 2015). A PPA program that monitors and responds to the social dynamism of PPAs such as landholder satisfaction, ownership changes, compliance and enforcement issues, and fluctuating landholder capacities, will be better suited to maintaining stewardship over the long term and delivering biodiversity benefits. We examine the factors that influence PPA stewardship and effectiveness in an Australian context.

1.2. PPA programs in Australia

Australia has among the most expansive and longest-running PPA programs in the world (Fitzsimons, 2015; Stolton et al., 2014), with a suite of mechanisms similar to other countries with well-developed PPA programs, such as New Zealand, South Africa, and the United States. The Australian PPA estate is comprised of land owned and managed by 1) non-profit conservation organizations; and 2) individuals on freehold and/or leasehold lands protected by conservation covenant (Fitzsimons, 2015). Conservation covenants are voluntary legally-binding in-perpetuity agreements, written into the land title deeds, between a landholder and a covenanting-administering organization, typically a state government agency or not-for-profit land trust, with legal authority to sign such agreements under legislation (Fitzsimons, 2006). Covenants are accompanied by plans of management, stewardship visits, and staff phone-calls (Fitzsimons and Carr, 2014). Breaches of covenant agreements are reportedly rare (Hardy et al., 2017). Despite the relatively longevity of programs and number of covenants there have only been a small number of published surveys of the views, perceptions and behaviors of covenants (e.g. Fitzsimons and Wescott, 2007; Halliday et al., 2012; Stephens et al., 2002).

In this research we aimed to 1) identify the levels of landholder satisfaction with Australian PPA programs and the factors that define it; 2) identify the challenges faced by landholders in managing their lands for biodiversity and in participating in a PPA program; 3) develop and advance a more holistic understanding of PPA stewardship and in so doing; 4) provide Australian conservation covenant programs guidance for planning and implementing their future monitoring, evaluation and program expansion activities.

2. Methods

2.1. Study region

Our study focused on PPA programs within the states of New South Wales, Tasmania, and Victoria, located in southeastern Australia (Fig. 1). Each state has its own legislation for managing species and ecosystems, but the national Environment Protection and Biodiversity Conservation Act 1999 also applies for threatened species or ecological communities listed as being of national significance. The region is comprised of a diverse range of ecosystems, including an internationally recognized biodiversity hotspot, the Forests of East Australia (Williams et al., 2011). The human population is approximately 7.7 million; 6.0 million and 0.5 million for New South Wales, Victoria and
Tasmania, respectively, and predominantly concentrated in urban areas. Dominant land-uses include irrigated and dryland cropping, small stock and cattle pastoralism, forestry and increasingly urban development (Australian Bureau of Statistics, 2010).

The Australian Land Conservation Alliance (ALCA; http://www.alca.org.au/) forms a national alliance of conservation organizations that either facilitate or implement PPA programs (conservation covenants as well as land acquisition and revolving funds). Across the study area, the Nature Conservation Trust (NCT)\(^1\) and Trust for Nature (Victoria) (TfN) manage PPA programs in NSW and Victoria, respectively. The Protected Areas on Private Land Program in Tasmania comprises a partnership between the Tasmanian Government Department of Primary Industries, Parks, Water and Environment (DPIPWE) and the non-government Tasmanian Land Conservancy (TLC) (Table 1; DPIPWE, 2017). DPIPWE holds and administers the covenant and TLC facilitates and supports the covenant, whereas NCT and TfN manage all covenant authority responsibilities. Other organizations that manage PPA programs in these three states (at the time of the survey in 2016) were approached but did not participate (NSW Government’s Office of Environment, Land, Water and Planning).

2.2. The Stewardship Functions Inventory

Functionalist, a concept from social psychology, theorizes that people have attitudes or motivations towards activities based on fulfilling multiple and potentially differing psychological ‘functions’ (Katz, 1960; Smith et al., 1956). In this study we used the Stewardship Functions Inventory (SFI), a psychometric instrument, comprised of Likert scales, that draws from functionalism, developed to assess the motivations that drive landholders to voluntarily participate in conservation activities on privately-owned land and their resulting level of satisfaction from participating (Selinske et al., 2015). It was based on the Volunteer Functions Inventory (VFI) used by Clary et al. (1998) to assess the motivations of people volunteering in diverse activities from healthcare (Omoto and Snyder, 2002) to recreational sports (Kim et al., 2010), and among different demographic groups such as college students (Gage III and Thapa, 2012) and retirees (Okun et al., 1998). The SFI is comprised of a total of 11 motivation subscales (Table 2), and one scale rating satisfaction, with a combined 58 items total. Five of the motivational subscales are drawn directly from the VFI and applicable to generic volunteer activities, six are novel, relating specifically to PPA programs.

2.3. The Landholder Survey

We adapted the wording of the questionnaire from Selinske et al. (2015), including the SFI items, with input from ALCA members to ensure relevance to the study region. Additionally, new questions requested by ALCA members and pertinent to Australian PPAs were included but are not reported in this paper. The questionnaire comprised both open-ended and closed questions exploring the landholder’s relationship with their land, their motivations for enrolling, their expectations, the benefits resulting from participating in the program, and their opinions of the support they receive from their covenanting organizations. A copy of the questionnaire is included in the Supplementary material.

The questionnaire was piloted in Australia on five landholders from New South Wales and Victoria and five staff members of covenanting organizations. Landholders were identified as potential participants using established email lists of the three state-based programs. Covenanting organizations distributed the questionnaire via emailed weblinks to Qualtrics survey software (Qualtrics, 2016; www.qualtrics.com) from September to mid October 2016 using the Dillman Tailored Design Method (Dillman et al., 2009). In NSW and Victoria, through NCT and TfN, one email request inviting landholders (NSW n = 128; Victoria n = 677) to participate in the survey was followed by three email prompts. Landholders who did not respond to emails and for which email addresses were unknown (519 in Victoria, 9 in NSW) were contacted once via a postal survey, but cost constraints restricted follow-up mailings. In Tasmania, only those landholders with known email addresses were contacted (n = 397), and those without email addresses (n = 417) were not contacted, potentially biasing the Tasmanian sample. Due to capacity constraints only two email reminders were sent to Tasmanian landholders, potentially providing a limited perspective of the Tasmanian covenantors.

2.4. Survey analysis

Quantitative data analysis included factor and scale reliability analyses using R Version 1.0.136 (R Development Core Team, 2016; https://www.r-project.org/) and the R Psych Package (Revelle, 2018; https://cran.r-project.org/web/packages/psych/index.html). Exploratory factor analysis (EFA) was used to confirm that subscale structures were appropriately aligned, ensuring each motivation subscale or satisfaction scale was a ‘cohesive’ construct. Only subscale items with factor loading values ≥0.32 were included (Tabachnick and Fidell, 2007). Each motivation and satisfaction subscale was tested for consistency and reliability in representing its intended construct. Despite the pervasive use of Cronbach’s Alpha (Cronbach, 1951), it is considered an inadequate measure of reliability (Zinbarg et al., 2005) and as such we applied the McDonald’s Hierarchal Omega (\(\omega\)) (McDonald, 1999) to determine coefficient scores and inter-item correlations. We calculated the means for each subscale. Testing for differences among groups of landholders and predictors of satisfaction was beyond the scope of the reported study and will be the subject of future analyses.

Responses to open-ended questions were coded and analyzed in NVIVO 11 qualitative analysis software (QSR International, 2016; http://www.qsrinternational.com) by two co-authors (MJS and NH).

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\(^1\)In August 2017 the Nature Conservation Trust was replaced by the NSW Biodiversity Conservation Trust.
### Table 1

The motivation functions of the Stewardship Functions Inventory. (adapted from Selinske et al., 2015).

<table>
<thead>
<tr>
<th>Motivations</th>
<th>Functions</th>
<th>Supporting literature</th>
</tr>
</thead>
<tbody>
<tr>
<td>Volunteer Functions Inventory</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Conservation Values</td>
<td>The individual enrolls in a covenanting program in order to express or act on the value of conservation</td>
<td></td>
</tr>
<tr>
<td>Understanding</td>
<td>The landholder is seeking to learn more about nature and how best to manage it on their land</td>
<td>Clary et al. (1998)</td>
</tr>
<tr>
<td>Social (Normative)</td>
<td>The covenanting program allows an individual to strengthen his or her social relationships and conform to social expectations</td>
<td></td>
</tr>
<tr>
<td>Ego Enhancement</td>
<td>A landholder can grow and develop psychologically through participation by protecting their land with a covenant</td>
<td></td>
</tr>
<tr>
<td>Ego Protection</td>
<td>The individual uses the covenanting program to reduce negative feelings, such as guilt, or to address personal problems. These feelings may be associated with an individual’s negative environmental impact</td>
<td></td>
</tr>
<tr>
<td>Stewardship Functions Inventory</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Stewardship Extension</td>
<td>A covenant offers a chance for a landholder to receive visits from and build a relationship with an extension officer</td>
<td>Moon and Cocklin (2011)</td>
</tr>
<tr>
<td>Stewardship Partnership</td>
<td>A landholder participates in the covenanting program to be part of a joint effort or larger movement to protect nature</td>
<td>Fitzsimmons and Wescott (2007); Cooke et al. (2012); Rissman and Sayre (2012)</td>
</tr>
<tr>
<td>Stewardship Incentives</td>
<td>A landholder is interested in the financial incentives available to covenant landholders (i.e. land tax exclusion, income tax reduction, or grants)</td>
<td>Miller et al. (2010); Moon and Cocklin (2011)</td>
</tr>
<tr>
<td>Social Network</td>
<td>The covenanting program gives the landholder an opportunity to expand their social network by meeting new people</td>
<td>Pasquini et al. (2010)</td>
</tr>
<tr>
<td>Place Attachment</td>
<td>The landholder enrolls in a covenanting program as a result of a strong emotional or spiritual relationship with the land often referred to as “a sense of place”</td>
<td>Cross et al. (2011); Farmer et al. (2011)</td>
</tr>
<tr>
<td>Business</td>
<td>The landholder has the goal of enhancing a business activity; which takes place on the covenanted property through marketing or protected area status</td>
<td>Selinske et al. (2015); Clements et al. (2016)</td>
</tr>
</tbody>
</table>

Responses were coded into multiple categories if answers raised several points. The iterative thematic analysis was discussed and confirmed within the research team and by partner organizations (Braun and Clarke, 2006; Kitchin and Tate, 2000). Kruskal-Wallis chi-squared test was used to test for non-response bias. We compared the answers of landholders that responded to the questionnaire in the first week to those that responded after the fourth reminder and compared Victorian landholders that did not respond to the mailed survey but responded to the posted survey (Groves et al., 2001).

### 3. Results

The response rate collectively for all states was 30.9% (n = 527). Thirteen of the postal surveys from Victoria were returned as incorrectly addressed. Our testing for non-response bias, revealed no interpretable bias between the two response groups. A majority (64.2%) of the responding landholders were 60 years of age or older, 47.1% of respondents were retirees and just over half (50.4%) resided on their covenanted land, indicating nearly half were absentee landholders (Table 2). Production landholders (those covenantors deriving incomes from the farming) made up 18.5% of respondents. Seventy-seven respondents (14.6%) were successor covenantors, who had either purchased covenanted land or inherited it from family. For 8.7% of landholders, the land associated with the covenant had been in their family for two or more generations. The average length of time that the covenants had been in place on the land varied across the states, with Victoria having the oldest covenants as a result of its longer running program (Table 2). Additional demographic and landholder data is available in the Supplementary material.

#### 3.1. Motivations to participate in the covenanting program

The SFI factor analysis (Supplementary material Table A1) and McDonald’s Omega (ωp) coefficient of reliability (Table 3) demonstrated high integrity of the scale structures and their a priori themes. Conservation Values, Place Attachment, Stewardship Partnership, Understanding and Social Norms were (in rank order) respondent’s most important motivations for joining a PPA program. The remaining motivations means had significantly lower ranks (μ ≤ 2.46). Thematic analysis of 420 qualitative responses revealed common motivations for respondent’s participation. The top motivation was to protect their land in perpetuity (95.0%; n = 399). Respondents joined the program to ensure ‘long term protection of the land’ and ‘to prevent the land being cleared after my death’. Many respondents also expressed a desire to restore the land they were protecting (36.2%; n = 152). One respondent in NSW expressed that they joined as an ‘opportunity to restore rainforest and other vegetation and improve wildlife habitat’. Other notable motivations were to maintain a spiritual or historical connection with their land (18.8%; n = 79); to receive financial assistance for management (16.9%; n = 71); and to receive technical assistance with land management (10.2%; n = 43).

#### 3.2. Benefits from participating in the covenanting program

Mentioned benefits of being involved in the covenanting program included protection of their land in perpetuity (32.2%; n = 147 of 456 respondents) or in one respondent’s words ‘[A] sense of security and knowing that my block will be protected after I am gone’, wellbeing benefits such as ‘a sense of pleasure and achievement’ (28.9%; n = 132 of 456 respondents), technical assistance in the form of ‘information and assistance’ and ‘on-going support, guidance and help we get from the staff involvement’ (24.1%; n = 110 of 456 respondents). Financial incentives such as management funding or tax rate rebates were seen as a benefit for 12.1% (n = 55 of 456) of respondents. While only 9.2% of respondents mentioned a social network as a benefit, when respondents were asked specifically if there was potential benefit to meeting with other covenantors 60.8% (n=273 of 449) believed there was, primarily to ‘exchange knowledge’, ‘share experiences’ and ‘pool resources’. Some respondents were partially dissatisfied with the benefits they received (or did not receive) from participation; with 11.7% (n = 62 of 456) stating they had received no benefits. Although some respondents did not clarify what benefits were expected just noting ‘to be honest we’ve received very little benefit at all’, others detailed their complaints: ‘there’s some financial reward, in the form of a rebate from the local council but we have not received [the rebate] and would prefer support from the [covenantee organisation] on a regular basis’.
Table 2
Landholder and conservation covenant program characteristics.

<table>
<thead>
<tr>
<th>State</th>
<th>New South Wales</th>
<th>Tasmania</th>
<th>Victoria</th>
</tr>
</thead>
<tbody>
<tr>
<td>Covenanting organization</td>
<td>Nature</td>
<td>DPIPWE/Tasmanian Land Conservancy</td>
<td>Trust for Nature (Victoria)</td>
</tr>
<tr>
<td>Year of first covenant</td>
<td>2005</td>
<td>1999</td>
<td>1986</td>
</tr>
<tr>
<td>No. landholders in program</td>
<td>110</td>
<td>814</td>
<td>1196</td>
</tr>
<tr>
<td>Surveys distributed</td>
<td>128</td>
<td>397</td>
<td>1183</td>
</tr>
<tr>
<td>No. of responses to survey</td>
<td>37 (online), 18 (post)</td>
<td>95 (online), 3 (post)</td>
<td>176</td>
</tr>
<tr>
<td>Response rate (%)</td>
<td>43.0</td>
<td>24.7</td>
<td>31.6</td>
</tr>
<tr>
<td>Avg. area of covenant (ha)</td>
<td>256.6</td>
<td>127.8</td>
<td>56.2</td>
</tr>
<tr>
<td>Avg. percentage of total property area owned that the covenant covers (%)</td>
<td>65.6</td>
<td>64.8</td>
<td>68.6</td>
</tr>
<tr>
<td>Avg. age of covenant (years)</td>
<td>6.4</td>
<td>9.2</td>
<td>13.1</td>
</tr>
<tr>
<td>Age of respondents (%)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>21–30</td>
<td>2.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>31–40</td>
<td>0.0</td>
<td>2.3</td>
<td>1.3</td>
</tr>
<tr>
<td>41–50</td>
<td>12.2</td>
<td>4.6</td>
<td>9.7</td>
</tr>
<tr>
<td>51–60</td>
<td>22.5</td>
<td>26.4</td>
<td>25.4</td>
</tr>
<tr>
<td>61–70</td>
<td>30.6</td>
<td>36.8</td>
<td>40.4</td>
</tr>
<tr>
<td>&gt; 71</td>
<td>32.7</td>
<td>29.9</td>
<td>23.3</td>
</tr>
<tr>
<td>Highest education level (%)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>&lt; Year 12</td>
<td>10.9</td>
<td>10.7</td>
<td>9.5</td>
</tr>
<tr>
<td>Year 12</td>
<td>8.7</td>
<td>6.0</td>
<td>7.2</td>
</tr>
<tr>
<td>TAFE Cert/Dip</td>
<td>10.9</td>
<td>14.3</td>
<td>19.0</td>
</tr>
<tr>
<td>Undergraduate</td>
<td>26.1</td>
<td>31.0</td>
<td>22.6</td>
</tr>
<tr>
<td>Postgrad diploma</td>
<td>19.6</td>
<td>16.7</td>
<td>15.0</td>
</tr>
<tr>
<td>Masters</td>
<td>13.0</td>
<td>14.3</td>
<td>16.7</td>
</tr>
<tr>
<td>Doctorate</td>
<td>6.5</td>
<td>7.1</td>
<td>7.8</td>
</tr>
<tr>
<td>Other</td>
<td>4.4</td>
<td>0.0</td>
<td>2.3</td>
</tr>
<tr>
<td>Income (%)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>No income</td>
<td>0.0</td>
<td>2.4</td>
<td>0.6</td>
</tr>
<tr>
<td>&lt; $18,200</td>
<td>6.1</td>
<td>5.9</td>
<td>8.0</td>
</tr>
<tr>
<td>$18,200–37,000</td>
<td>18.4</td>
<td>25.9</td>
<td>19.0</td>
</tr>
<tr>
<td>$37,001–80,000</td>
<td>28.6</td>
<td>28.2</td>
<td>30.9</td>
</tr>
<tr>
<td>$80,001–180,000</td>
<td>34.7</td>
<td>20.0</td>
<td>21.5</td>
</tr>
<tr>
<td>&gt; $180,001</td>
<td>6.1</td>
<td>7.1</td>
<td>9.7</td>
</tr>
<tr>
<td>Prefer not to answer</td>
<td>6.1</td>
<td>10.6</td>
<td>10.3</td>
</tr>
<tr>
<td>Landholders that derive income from covenanted land (%)</td>
<td>7.4</td>
<td>8.5</td>
<td>8.9</td>
</tr>
</tbody>
</table>

* If a covenant had multiple owners, all were sent a survey.

3.3. Expectations of the covenanting program

Landholders considered the responsibilities of covenanting organizations as primarily providing technical advice (59.7%; n = 261 of 437 respondents), such as for management strategies and advice on applying for financial assistance, and monitoring and enforcing the compliance of covenants and management agreements (35.0%; n = 153 of 437 respondents). Landholders felt that it was the duty of the covenanting organization ‘to ensure that we observe the terms of the covenant and that it [protection] will continue after my lifetime’. Additionally, the desired number of visits from management organization staff for 65.0% of the respondents was between one visit per year and a visit every three years.

3.4. Landholder capacity to meet covenant management agreements

While a majority of respondents (71.1%; n = 328 of 461) reported being able to meet the management requirements of their agreement, including 43.0% who felt they invested beyond the agreement requirements, challenges were faced by many of the respondents. Of the 461 responses, 44.7% expressed concerns regarding the effectiveness of their management efforts. Hindrances included the financial costs of management (12.5%; n = 58), time constraints (8.0%; n = 37), age (18.0%; n = 83), and a general feeling of a lack of self-efficacy (9.8%; n = 45). Some respondents felt that their limited knowledge of fire and invasive species management hindered the efficacy of their land management (7.5%; n = 35). Of the respondents, 9.3% (n = 43) conveyed facing multiple management challenges. Some landholders were also apprehensive with the management capacity of future owners of their covenant. In response to the statement ‘I am concerned about how future owners will manage the natural values on my covenanted land’, 16% (74) of landholders either strongly agreed or agreed.

3.5. Satisfaction with the covenanting program

Most respondents (92%) rated their satisfaction of the experience in the covenanting program highly (μ = 4.1), although 8% had mean satisfaction scores of 3.00 or less, expressing dissatisfaction. Of the 77 landholders that purchased or inherited a covenant, 17% were dissatisfied, compared to just 6% of original covenantors. From our thematic analysis we found that satisfaction was connected to a number of factors. Overall respondents were satisfied with their covenanting arrangements (92.6%; n = 426 of 460), their ability to meet management goals (71.1%), the annual number of interactions with their covenanting organization (61.0%) and the assistance they received from the covenanting organization (58.9%). Some respondents that expressed dissatisfaction with the support provided by the covenanting organization (38.0% of n = 455) did so because they felt the assistance was limited, it declined since enrolling in the program, or that they had not received assistance since the covenant was in place. Over one-third of respondents (39.0%) were dissatisfied with the number of extension officer visits, which were wanted for obtaining technical advice and for compliance monitoring: ‘to ensure that we observe the terms of the covenant and that it will continue after my lifetime’. Based on these responses and some landholder’s frustration with the inability to manage their land in the way they expected, a small proportion of respondents (4.97% of 465 responses) would not re-enroll their land in a covenanting program if they had that option.

4. Discussion

Privately Protected Areas in Australia have seen dramatic growth in recent decades for a variety of reasons, including governments seeking to meet national and global protected area targets, and landholders acting to conserve the nature on their land (Fitzsimons, 2015; Hardy et al., 2017). Ensuring that PPA programs foster ongoing commitment to stewardship from landholders to manage their lands, contribute to the integrity of the contract, and remain engaged in the program, is
central to securing ongoing conservation outcomes. In surveying PPA landholders, we found a number of factors that likely influence their ongoing stewardship, including 1) motivations for participating, 2) perceived benefits from participating, 3) capacity to manage their covenant, 4) expectations of the covenanting organization, and 5) resulting satisfaction levels. Here we discuss the implications of these factors for program design of PPA programs and stewardship on private lands.

We found through qualitative and quantitative analyses that landholders in NSW, Tasmania, and Victoria are motivated to place in perpetuity conservation covenants on their land for multiple reasons. Overwhelmingly, landholders expressed that the permanent protection of their land was the main driver for participating in the covenant program. While the SFI did not explicitly measure a permanent protection motivation, we did find the strongest SFI motivations were the fulfillment of Conservation Values, and Place Attachment, which are both associated with landholders’ conservation identity and likely the ‘latent’ or underlying motivation behind the expressed desire for permanent protection. Additionally, landholders want to partner with organizations such as Trust for Nature (Victoria), NSW Nature Conservation Trust or Tasmanian Land Conservancy to feel part of a larger movement contributing to a shared endeavor (Stewardship Partnership SFI motivation). Landholders are also motivated to participate to understand more about their land and how best to manage it, and because of social norms, such as meeting the expectations of friends and family (Social Norm SFI motivation).

For landholders, the greatest benefit from participating was the protection of their land, aligning with their strongest motivation. Psychological wellbeing, financial assistance for land management, land-tax relief, and interaction with a stewardship officer did not feature as strong motivations but ranked higher among mentioned benefits. As stated previously, this may reflect the latent nature of certain motivations or potentially the importance of expected and/or unexpected benefits ancillary to the main drivers of participation. Delivered benefits that are expected as part of a covenant program, such as technical and financial assistance in land management, are likely to increase landholder satisfaction with the program. Covenantor social networks were not a predominant motivation, nor a mentioned benefit of the program, likely as a result of a lack of organized interaction between landholders. Meeting with other covenantors can be a positive PPA program benefit (Horton et al., 2017) and given the interest expressed by respondents to regularly meet with other covenantors, developing social networks is likely to result in more satisfied and potentially higher capacitated participants.

Although satisfaction rates are high, a number of landholders surveyed expressed dissatisfaction, corroborated by qualitative analyses and the SFI. These landholders felt they were not receiving the support they expected from their covenanting organization. Similar to findings by Stephens et al. (2002), landholders rated their interaction with extension officers as highly important and many desire an increased amount of visits, with some landholders dissatisfied by the very few (or in some cases no) visits received since enrolling their lands. Further, limited interaction with the covenanting authority results for some in a perceived weakness of enforcement monitoring, concerning landholders who are not confident that the next owner will protect or manage their covenanted land according to its conservation values and the agreement made with the covenanting organization. This set of results highlights the importance of securing resources that enable regular landholder visits for covenanting programs in Australia, and likely PPA programs worldwide.

Given that the perpetual nature of covenants and their obligations on future landholders is a key motivation for participation, and that covenant landholders risk financial and legal implications by not complying with their covenant agreement, it is unsurprising that the numbers of reported covenant breaches are low (Hardy et al., 2017). Nonetheless, the ability of landholders to meet the management goals for their land impacts conservation outcomes. Landholders see their role to be undertaking conservation management activities, and for covenanting organizations to enable and facilitate this conservation to take place. A sizable minority of landholders (44.7%) feels constrained in their ability to manage their covenant, as a result of age, financial costs, or time commitments resulting in a lack of self-efficacy to manage the PPA for conservation. This corresponds to but differs slightly from the findings of Fitzsimons and Carr (2014) in their surveys of conservation covenanting organizations: 5 of 7 of those organizations specified the greatest barrier to effective management was the lack of covenantor time, while 3 of 7 considered a lack of resources to be a barrier. Landholders identified invasive species and fire management as areas where they need greater management support, consistent to findings elsewhere in Australia (Fitzsimons, 2015; Halliday et al., 2012).

The majority (64.2%) of PPA landholders in this study are over the age of 60, which presents substantial challenges for long-term PPA stewardship. Many land management activities are relatively physical, and aging landholders may require additional assistance. While many landholders in the study indicated they have made large financial conservation investments in their property, their ongoing financial capacity will likely reduce as they retire and some will be unable to maintain this level of investment into their covenants. This is especially problematic where financial assistance is not made available for expensive works such as fence repairs or continued invasive species control.

Inevitably, through being sold or inherited, some covenanted lands will end up in the ownership of people who may not prioritize the management of nature. The results indicate that successor landholders made up a large portion of dissatisfied respondents, similar to the findings of Fitzsimons and Carr (2014) where successor covenantors were suggested to be less enthusiastic towards covenants and covenant management. While successor covenantors were only a small proportion of respondents in this study, subsequent discussions with covenanting organizations leading up to and during property handover, and additionally, actively encouraging conservation-minded buyers through targeted property advertising (see Hardy et al., 2018; www.trustfornature.org.au/land-for-sale/; http://tasland.org.au/properties-for-sale/).

PPA initiatives worldwide can achieve a great deal by engaging increasingly receptive private landholders. However, by increasing the number and extent of PPA there is risk of overstretching covenanting organizations through servicing landholders that require assistance and/or carrying out important functions such as monitoring land for management and compliance. Unaddressed, landholders who are dissatisfied with the program, unable to implement management tasks, or successor landholders that do not wish to comply with agreements, are risks to the integrity of covenants that could result in suboptimal biodiversity conservation outcomes.

Our findings are supported by research in other contexts and program types. In South Africa, PPA landholders ranked conservation values, place attachment and social learning, as their top motivations for participating and satisfaction was dependent on their interactions with an extension officer and perceived efficacy of the program (Selinske et al., 2015). Among Texas easement-holders, program design, landholder motivations, relationship with easement organization, and...
satisfaction with the program influenced the type and efficacy of management activities on easements (Stroman and Kreuter, 2015, 2016). Rissman and Sayre (2012) found that a strong social network between the landholders and easement organizations strengthens conservation outcomes, specifically management activities. In New Zealand, management effort and expenditure towards covenants were influenced by landholder socioeconomic characteristics, and landholders expressed concern about their ability to manage the land due to weed control expenditures and age (Scrimgeour et al., 2017). Furthermore, it has been found in the US that successor landholders of easements have undermined and legally challenged title-deed restrictions (Collins, 2000; Rissman and Butsic, 2011), potentially weakening their long-term societal value (McLaughlin, 2005).

Similarly, research in tender-based private land conservation programs found that both production and amenity landholders participate in private land conservation as a result of conservation values (Cooke and Corbo-Perkins, 2018; Moon et al., 2012). Personal finances impact the ability of landholders to actively manage their land for biodiversity (Moon and Cocklin, 2011) and engagement with an extension officer was reflected as one of the most enjoyable parts of these programs (Burmeister et al., 2006; Selinske et al., 2017). Multiple social-psychological and economic factors influence the continued management of land enrolled in tender-based approaches, and short-term and permanently protected private land conservation programs (Dayer et al., 2018; Farmer et al., 2017). As a result, consideration of continued engagement with landholders post-enrollment in tender-based arrangements is required and assumptions of continued management or protection of private lands and associated conservation outcomes should not be made without investigation.

Our findings are likely applicable and of interest to existing PPA programs in countries such as Brazil, New Zealand, South Africa, and the United States, as well as those that are new or in development. Despite the likely diversity of landholders and motivations, in many contexts it is probable that landholders, absent of large financial incentives, are motivated to participate in PPA programs by conservation values/attitudes and place attachment, resulting in a desire to preserve the landscape for biodiversity and cultural values. However, as we found in this study, once these primary motivations are met they may not be the main factors influencing satisfaction. Further research is needed to explore drivers of participation and factors contributing to continued management and satisfaction, beyond US easement holders and Australian covenantors (Simao and Cardoso Coelho de Freitas, 2018).

In Australia and South Africa, engagement with a stewardship officer is an important factor of landholder satisfaction (Selinske et al., 2015; this study), and this may also be the case elsewhere. Landholders see themselves in partnership with land trusts to achieve conservation outcomes, and are seeking guidance and support to manage their land in best practice (see Mitchell et al., 2018). Given this, it would appear beneficial for PPA programs currently under development to invest in regular stewardship outreach – whether through site visits, via telephone, or electronic communication. Where capacity for outreach is limited, then the expectations of participating landholders will need to be managed. Below we outline five recommendations to engender PPA stewardship through PPA program design and delivery:

1. **Evaluate the motivations and expectations of new or interested PPA participants.** Meeting expectations and catering to motivations will increase the likelihood of fostering satisfaction with PPA programs (Knight et al., 2010). If it is not possible to meet all landholder’s expectations then making clear what the capacity of the organization is, and what it can deliver, prior to enrollment may help ameliorate dissatisfaction.

2. **Regular surveys of existing landholders to gauge satisfaction with program, needs, and capacity to manage.** Identify challenges that landholders face in meeting management goals. Determine if challenges are localized among certain groups or communities of landholders or systemic (Selinske et al., 2015; Stroman and Kreuter, 2015). This should be done in addition to regular site visits with landholders.

3. **Integrate information derived from surveys into program design and delivery.** Target capacity building based on needs for specific groups such as older residents (financial support and support for physically demanding work) (Mendham and Curtis, 2010); absentee landholders (time and information deficient) (Bond et al., 2018) and for others pursuing specific highly technical work (e.g. fire management) (Raymond et al., 2015).

4. **Build formal and informal landholder stewardship networks.** Facilitating the development of social networks (e.g. through field days, events) will promote social learning, knowledge sharing and adaptive management, reinforce positive social norms, provide management assistance, and build community capacity for conservation. This can be driven by the landholders themselves (see Conservation Landholders Tasmania https://www.landcaretas.org.au/1695 and Graham and Rogers, 2017).

5. **Engage in the land transfer process.** Work with relevant interested government agencies, NGOs, financial institutions and real estate agents to develop a service that facilitates the transfer of ownership of PPA lands. This service could help connect conservation-minded buyers with PPA landholders, educate buyers of the conditions associated with owning a PPA, and develop the market for PPAs over time (Gruver et al., 2017; Markowski-Lindsay et al., 2017).

5. **Conclusions**

Assessing the management effectiveness of protected areas and protected area networks, including PPAs, is increasingly important (Hockings et al., 2006). In addition to ecological and compliance monitoring (Fitzsimons and Carr, 2014; Hardy et al., 2017), it is equally important for PPA initiatives to consider the social dimensions and outcomes of PPA programs. Effective PPA program design is founded upon understanding the reasons why stakeholders get involved and how their commitment is engaged and sustained (Lindsay, 2016). Within this, monitoring the perspectives of PPA landholders can provide PPA organizations insight into the social dimensions of their programs, helping to establish return on investment and reduce institutional risk.

Using behavioral and psychological theory can help inform the assessment of landholder perspectives for PPA programs, and in conservation more broadly (Selinske et al., 2018). The SFI instrument shows promise as a tool for assessing landholder commitment and determining the dimensions that underpin it, across multiple contexts (see Selinske et al., 2015). In applying the SFI to the Australian conservation covenant context we find that landholder satisfaction of covenants and covenanting programs is instrumental in ensuring stewardship, likely influencing biodiversity outcomes on covenanted land. While most landholders surveyed were satisfied with their participation in a covenanting program, this satisfaction is dynamic between individuals and across generations, presenting challenges to the long-term effectiveness of covenant (and similar) programs. Incorporating the monitoring of landholders into the design and implementation of PPA programs will enable conservation organizations to respond to landholder needs, and help shape the ongoing effectiveness of conservation efforts on private land.

**Supplementary data to this article can be found online at** [https://doi.org/10.1016/j.biocon.2018.11.026](https://doi.org/10.1016/j.biocon.2018.11.026).**

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Moving from reactive to proactive development planning to conserve Indigenous community and biodiversity values

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A R T I C L E   I N F O

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Cultural impact assessment
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Native title
INDIGENOUS land rights
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NORTHERN Australia
DEVELOPMENT by Design
CONSERVATION Action Planning
Avoid
Minimize
Restore
Offset

A B S T R A C T

There is increased awareness of the need to balance multiple societal values in land use and development planning. Best practice has promoted the use of landscape-level conservation planning and application of the ‘mitigation hierarchy’, which focuses on avoiding, minimizing or compensating for impacts of development projects. However, environmental impact assessments (EIA) typically focus in a reactive way on single project footprints with an emphasis on environmental values and specifically biodiversity. This separation may miss opportunities to jointly plan for and manage impacts to both environmental and social values. Integrated approaches may have particular benefit in northern Australia, where Indigenous people have native title to as much as 60% of the land area and cultural values are closely linked with natural values. Here, we present a novel framework for integrating biodiversity and cultural values to facilitate use in EIA processes, using the Nyikina Mangala Native Title Determination Area in the Kimberley, Western Australia, as a case study. We demonstrate 1) how social and cultural values can be organized and analyzed spatially to support mitigation planning, 2) how social, cultural, and biodiversity values may reinforce each other to deliver better conservation outcomes and reduce social conflicts, and 3) how this information, in the hands of Indigenous communities, provides capacity to proactively assess development proposals and negotiate mitigation measures to conserve social, cultural, and biodiversity values following the mitigation hierarchy. Based on values defined through a Healthy Country Planning process, we developed spatial datasets to represent cultural/heritage sites, freshwater features, common native animals and plants represented by biophysical habitat types, and legally-protected threatened migratory species represented by potential habitat models. Both cultural/heritage sites and threatened species habitat show a strong thematic and spatial link with freshwater features, particularly the Fitzroy River wetlands. We outline some of the challenges and opportunities of this process and its implications for the Northern Australia development agenda.

1. Introduction

Large-scale development projects profoundly transform environments, communities, cultures and economies, and often generate social conflict (Hilson, 2002; Bridge, 2004; Hanna and Vanclay, 2013; Franks et al., 2014). These types of development will continue to expand as global population and consumption increase (Oakleaf et al., 2015). Environmental licensing processes, such as Environmental Impact Assessment (EIA), play a critical role in limiting impacts from development projects to both the environment and the affected communities. In most countries, developers are required to get an environmental license before development activities can begin, and EIA has been legally adopted in almost all countries in the world (Morgan, 2012; Villarroya et al., 2014). The scientific community has responded to this requirement with decades of research establishing the mitigation hierarchy and best practices for mitigation of impacts to biodiversity (e.g.
Kiesercker et al., 2010; Maron et al., 2015; Tallis et al., 2015), as well as conventions and systems for maintaining and sharing biodiversity information (e.g., Dunn and Weston, 2008; Lewis et al., 2008). When applied in the earliest stages of the decision-making process, EIAs can become important project planning instruments, providing information describing the consequences of specific development activities in a way that can inform approval decisions and design mitigation measures.

Since EIA is the most developed policy instrument, backed by a legal framework in many countries, it is increasingly also used to assess the social and economic impacts of planned interventions. Values considered by the EIA processes include primarily environmental values, with a focus on biodiversity. However, there is growing recognition that impact assessments and mitigation requirements should include social and cultural values with systematic frameworks and standards (Arce-Gomez et al., 2015; Vanclay et al., 2015; Partal and Dunphy, 2016). There are already International standards that call for the conservation of cultural and social values, including the UN Declaration on the Rights of Indigenous Peoples, UN Sustainable Development Goals, and the International Finance Corporation Performance Standards (IFC, 2012), and require assessment of risks and impacts to cultural values. Additionally, as recognized by the Millennium Ecosystem Assessment (2005), while society’s demand for cultural services has continued to grow, the capability of ecosystems to provide cultural benefits has been significantly diminished in the past century. Ecosystem services are generally classified by type as provisioning, regulating, habitat/supporting, and cultural (Millennium Ecosystem Assessment, 2005; TEEB, 2011). Cultural ecosystem services (CES), defined as the non-material benefits of ecosystems and human-environment interactions, are often missing from management policy (Chan et al., 2012, 2016; Pascua et al., 2017).

In recognition of the rights of people to maintain their social and cultural identity, the concept of Free, Prior and Informed Consent (FPIC) has been established as a specific right of Indigenous peoples and is recognized in the United Nations Declaration on the Rights of Indigenous Peoples, the United Nations Universal Declaration of Human Rights, the International Labour Organization Convention 169 (Indigenous and Tribal Peoples Convention, 1989), and the Convention on Biological Diversity. FPIC is intended to enable communities to give or withhold consent to a project that may affect them or their territories and to negotiate the conditions under which the project will be designed, implemented, monitored and evaluated. A key component of the FPIC framework is that consent is sought sufficiently in advance of any authorization or commencement of development operations (Hanna and Vanclay, 2013; Vanclay et al., 2015). But like EIA, FPIC is typically a reactive process not initiated until a government entity or company informs an Indigenous community of their intention to develop within their territory. As a result, the typical project review process does not allow adequate assessment of impacts to social and cultural values because of the time, data, and technical capacity required.

Efforts to conserve biodiversity globally have developed best practices and data systems that facilitate effective impact assessment, such as criteria for threatened species designations based on rarity and vulnerability (Ricketts et al., 2005; Langhammer et al., 2007; IUCN, 2017). These have been widely adopted in EIA law and policy (Villarroya et al., 2014) and are recognized by developers and lenders (IFC, 2012), with resulting benefits for biodiversity conservation. Similar constructs to organize information to inform mitigation of impacts to social and cultural values have not been universally adopted. In many landscapes, biodiversity and cultural/social values are intricately related (Altman, 1987; Asafu-Adjaye, 1996; Garnett et al., 2009; Hill et al., 2013; Moorcroft et al., 2012). The decision-making process will benefit from a more integrated approach, particularly for developments impacting Indigenous communities where cultural values are often of great importance.

Impact assessment that considers environmental, social and economic values requires an integrating framework. In many cases, environmental impact assessment and social impact assessment have operated in separate realms. To date, few unified conceptual frameworks exist to guide the standardized integration of biodiversity and social/cultural values into environmental impact assessments or development proposals, despite Indigenous people owning or having legal title to a large portion of the world’s lands and water (Oxfam, 2016; Willy et al., 2017). Geneletti (2015) proposed a conceptual framework for integrating ecosystem services into strategic environmental assessments. Tallis et al. (2015) proposed a framework for integrated biodiversity and ecosystem services mitigation. Pascua et al. (2017) developed and demonstrated a framework for eliciting place-based CES. Principles and guidance exists for how to include social and cultural values in EIAs (Vanclay, 2003; Vanclay et al., 2015; Arce-Gomez et al., 2015) and in the specific context of ecosystem services (Karrass, 2016), but no systematic approach or analytical precedent for integrating cultural values with biodiversity has been proposed.

Therefore, we see a unique opportunity to advance mitigation for both biodiversity and cultural values jointly, to evaluate and demonstrate: 1) how social and cultural values can be organized and analyzed spatially to support proactive mitigation planning and management decisions, and how this can enable EPIC for Indigenous communities; and 2) how cultural/social and biodiversity values may reinforce each other to deliver effective conservation outcomes that address cumulative impacts at landscape-scales and that better account for social impacts. Here, we outline a method for incorporating biodiversity and cultural/social values into a development planning process, using a case study on Indigenous land in northern Australia. The result is a framework for mapping community-defined social, cultural, and biodiversity values to support EIA by enabling proactive impact analysis and informed negotiation of development proposals. The framework provides data and capacity to an Indigenous community to proactively assess development proposals and negotiate mitigation measures to avoid, minimize, and offset impacts following the mitigation hierarchy.

This framework is novel in two ways. First, it integrates spatial data representing social, cultural, and biodiversity values to enable impact analysis. Second, it provides this information directly to the Nyikina Mangala community and their aboriginal corporation, i.e. the Registered Native Title Body Corporate (RNTBC). As such, we expect that it will improve EIA processes by enabling proactive, informed assessment and negotiation of development plans on their native title lands. We discuss strengths and challenges to the process and applicability to other regions.

1.1. Background

Indigenous land management in Australia, often called ‘Caring for Country’, includes a wide range of environmental, natural resource and cultural heritage management activities undertaken by Indigenous individuals, communities and organizations. Resource use over more than 60,000 years occurred in accordance to the traditional and geographic patterns of the land, based on holistic relationships between Indigenous people and their customary land estates—or ‘Country’. This has resulted in close linkages between cultural and environment values (Altman, 1987; Asafu-Adjaye, 1996; Hill et al., 2013).

Traditional Owners hold native title rights to approximately 32% of Australia’s total land area, and as much as 60% of northern Australia, through Native Title Determinations as of March 2018 (National Native Title Tribunal, 2018). Native title is the recognition in Australian law that some Indigenous people continue to hold rights to their land and waters that are based on their traditional laws and customs. The Native Title Act 1993 (NTA) provides a system for the recognition and protection of native title rights and for its co-existence with other land-use interests. The Australian Indigenous estate has high national environmental significance and includes some of Australia’s highest conservation priority lands and a diverse range of
intact ecosystems (Altman et al., 2007).

Australia’s northern tropical savannas are considered the largest intact savanna in the world (Woinarski et al., 2007), with high endemism and globally-significant biodiversity (Carwardine et al., 2011, 2012; Pepper and Keogh, 2014), and occupy 99% of their original extent (Woinarski et al., 2011; Bradshaw, 2012). Following European settlement, changes in land-use and subsequent changes in fire regime and introductions of invasive species and novel disease modified significantly the composition and structure of the savannas (e.g., Woinarski et al., 2011). Today, major land uses include extensive pastoral activity, conservation management on Indigenous and public land (including traditional fire management) (e.g. Russell-Smith et al., 2009, 2015; Walton and Fitzsimons, 2015), and smaller areas of mining and irrigated agriculture.

1.2. Study area

The study area follows the boundaries of the Nyikina Mangala Native Title Determination (NTD), an area of approximately 26,100 km² that contains the Lower Fitzroy River and delta and the lower quarter (22%) of the Fitzroy River watershed. The Walalakoo Aboriginal Corporation, the Registered Native Title Body Corporate (RNTBC), was established to represent Nyikina and Mangala Traditional Owners interests and Native Title rights over this area (National Native Title Tribunal, 2014). Here, the Nyikina Mangala community faces a convergence of the issues described above that relate to integrated analysis and decisions about protection and management of environmental and cultural values in the face of existing and emerging development pressures. Indigenous rights holders face similar issues across northern Australia (Joint Select Committee on Northern Australia, 2014). The NTD lies on the southwestern side of the Kimberley Tropical Savanna Ecoregion (Olson et al., 2001) and across two IBRA biogeographic regions (Thackway and Cresswell, 1995; Environment Australia, 2000): Dampierland and the Great Sandy Desert.

The development and improved agricultural productivity of Northern Australia is the focus of multiple State/Territory and Australian government initiatives that aim to double agricultural output over the next 20 years (Joint Select Committee on Northern Australia, 2014). To achieve this goal, the Australian Government suggests new and expanding agricultural projects across 400,000 ha of land (Australian Government, 2015), mirrored by State-funded programs (e.g. Department of Primary Industries and Regional Development, 2017). Given rich mineral and petroleum resources, northern Australia’s mining and petroleum developments are expected to expand and will continue to provide a large percentage of Australia’s resource exports (Joint Select Committee on Northern Australia, 2014). If undertaken, these development proposals have implications for biodiversity and the ecosystem services of the largely natural landscapes in northern Australia (Morán-Ordóñez et al., 2017), as well as for cultural and social values of people that manage or depend on these landscapes (North Australian Indigenous Experts Panel, 2012).

2. Methods

This study began with a systematic definition of values by traditional owners in the Walalakoo Healthy Country Plan (WAC, 2017), a cultural and natural resource management plan that follows the
Healthy Country Planning (HCP) methodology. Based on this information, the community defined spatial priorities for avoiding development impacts. Last, we organized the spatial datasets in an information system to support community resource management decisions, development planning, and impact mitigation.

Healthy Country Planning is an adaption of Open Standards for the Practice of Conservation (Schwartz et al., 2012), a globally recognized planning framework that guides community and conservation groups through a multi-step participatory process for the development of an adaptive management plan (Carr et al., 2017). Through the HCP process, the community defines conservation values within a participatory planning framework. This facilitates the development of a structured understanding of their vision, values, threats and their interactions. The Healthy Country Planning methodology has been widely adopted throughout Indigenous Australia for the development of management plans for Indigenous Protected Areas and other Indigenous Land Management Initiatives (e.g. Moorcroft et al., 2012; Jupp et al., 2016; Carr et al., 2017; Austin et al., 2017, 2018).

The first step in the HCP process is to engage the community and define values or targets. The Nyikina Mangala community defined a set of seven natural, cultural, and socio-economic targets that collectively represent Nyikina and Mangala people’s values and vision for Healthy Country (See Table 1). Following the Open Standards for the Practice of Conservation (Schwartz et al., 2012), all HCP target definitions include key ecological attributes in terms of viability and integrity that include the ecosystem services provided. In terms of ecosystem service categories defined by the Millennium Ecosystem Assessment (2005) and TEEB (2011), all targets provide CES, and several targets also provide provisioning, regulating, and habitat/supporting services. To improve decision-making and the EIA process, we developed spatial datasets to represent and integrate social/cultural and biodiversity targets in an impact assessment framework. A detailed data management and intellectual property agreement was developed prior to gathering and collating information for the study.

To facilitate use in EIA processes we developed spatial datasets to represent cultural, social, and biodiversity values of the Nyikina Mangala community across the Native Title Determination (NTD), specifically four targets defined by the HCP: Cultural and Heritage Sites, Freshwater Places, Native Animals, and Bush Tucker/Bush Medicine Plants. The community defined threatened species protected by national and state legislation as nested targets within the target groups Native Animals and Bush Tucker/Bush Medicine Plants, in accordance with their traditional view of country. However, threatened species are typically addressed independently by legal regulations and mitigation requirements. For the purpose of this study, we describe cultural/social values and threatened and endangered species separately and analyze the relationship between them. This allows us to assess the additionality of listed threatened species to the larger range of culturally important values.

2.1. Cultural/heritage sites

The NTD contains hundreds of sites with significance to Nyikina Mangala lore and culture. These sites range from artefacts and rock art to ceremonial sites to physical features attached to traditional stories. We compiled a database of the locations and attributes of 663 sites identified in 18 surveys between 1983 and 2015, including sites in the register maintained by the Western Australian Department of Aboriginal Affairs (DAA). This dataset includes only survey records. The spatial pattern of site records is largely determined by survey effort, and areas without survey records may contain un-recorded sites.

To facilitate use of this cultural spatial data in EIA processes, the community working group defined areas to avoid development as a 2 km buffer around each cultural/heritage site. The 2 km zone is a placeholder pending a site survey for any development project. Development proposals that go forward must conduct site surveys to redefine the protection zone around each cultural/heritage site based on the specific characteristics of the site and the surrounding landscape.

2.2. Freshwater features

The freshwater places identified by the HCP include the Fitzroy River and tributaries, their floodplains and riverine wetlands, as well as springs and other wetlands and waterbodies occurring across the NTD and associated native flora and fauna. We mapped and classified these as four types of features: floodplains of the Fitzroy River and major tributaries, riparian areas of smaller tributaries, large water bodies and wetlands, and smaller ephemeral water bodies (details in Appendix 1). A national surface hydrology dataset (Geoscience Australia, 2015) delineates major floodplains, water bodies and wetlands at 1:250,000. Permanent and semi-permanent water bodies are critically important for Indigenous subsistence livelihoods, cultural heritage, and biodiversity (Jackson and Robinson, 2009) but locations of those water bodies are not mapped consistently. To address this data gap we delineated the floodplains and riverine wetlands of smaller tributaries with a topographic model (Smith et al., 2008) derived from a digital elevation model (Geoscience Australia, 2011; Gallow et al., 2011) at 1 arc-second (30 m) resolution, and mapped other small and ephemeral water bodies with a supervised multispectral classification of Landsat 8 OLI imagery (USGS, 2015) collected April 2015. The community working group defined freshwater protection zones to avoid development that consist of the floodplains and riverine wetlands of the Lower Fitzroy River, the Fraser Rivers, and their major tributaries that lie within the NTD.

2.3. Plants and animals identified for cultural-socio-economic purposes

‘Native animals’ include many common animal species that are valued for hunting. ‘Bush tucker/bush medicine plants’ also include many common plants species that are gathered for food, medicine, utensils, arts/crafts, and fuel. The distribution of common animals and plants generally follow patterns of biophysical habitat. To map the general distribution of common animals and plants, we developed a biophysical habitat classification (Fig. 2) that defines eleven biophysical habitat types across the Fitzroy Basin analysis area, including the freshwater features mentioned above. The classification typology is based on biogeography, landforms, vegetation structure, and surface hydrology (Appendix 1). The resulting mapped biophysical classification is a reasonable proxy for the distribution of common, widespread species and represents landscape-level environmental gradients and the physical template for broad scale processes necessary to maintain habitat (Hunter et al., 1988; Groves et al., 2002). However, the biophysical units will not capture the distribution of rare or sparingly-distributed species or species with habitat requirements that are not well-represented by the biophysical units. As such, the biophysical habitat classification also functions as a coarse filter for biodiversity, following

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Table 1

<table>
<thead>
<tr>
<th>List of targets defined in the Healthy Country Plan</th>
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<tbody>
<tr>
<td>1. Nyikina Mangala Lore and Culture: Language, dance, song, stories, ceremony, customs</td>
</tr>
<tr>
<td>2. Cultural and Heritage Sites: Rock-art, burial sites, massacre sites, old camping places, artefact scatter, old workshops and ceremony sites</td>
</tr>
<tr>
<td>3. Freshwater Places: Fitzroy River, springs, wetlands, creeks, billabongs, fish and birds, bush-fruit / medicine plants along the river</td>
</tr>
<tr>
<td>4. Native Animals: traditional food-sources and threatened and endangered animal species</td>
</tr>
<tr>
<td>5. Bush Tucker / Bush medicine Plants: traditional plants used for foods, medicine and tools</td>
</tr>
<tr>
<td>6. Right Way Fire Management: Early Dry Season burning implemented by Traditional Owners</td>
</tr>
<tr>
<td>7. Being Strong on Country: Being in control of country and being able to gain livelihoods from Nyikina Mangala country</td>
</tr>
</tbody>
</table>
a widely used coarse filter/fine filter strategy for conservation planning (Hunter, 1991; Noss, 1996; Groves, 2003), representing a major component of biodiversity: common native animals, plants, and ecological communities.

2.4. Species protected by state and national regulations and international agreements

Species listed as threatened or priority by state and national legislation that occur in the NTD area include 32 animals - 9 mammals, 15 birds, 6 fish, and 2 reptiles (DoE, 2016; WA DPaW, 2016a) and 19 plants (WA DPAW, 2015) as well as 18 migratory shorebirds protected by international agreements (DoE, 2015). State legislation also protects 3 threatened and priority ecological communities that occur in the NTD along the Lower Fitzroy River and have been designated and mapped by WA DPAW (2016b). We defined the threatened animals and migratory shorebirds as focal biodiversity targets, listed in Appendix 2, and developed spatial models of potential suitable habitat based on habitat definitions in literature and existing spatial data compiled in the biophysical spatial model. Because observation data for all these species is absent or very limited, we were not able to develop species distribution models derived from occurrence data. Instead, we developed models of potential suitable habitat for 22 threatened species (6 mammals, 11 birds, and 5 fish) and one model to represent the Lower Fitzroy riverine and estuarine wetlands used seasonally by the 19 migratory shorebirds. Source datasets and method details are listed in Appendix 2. The habitat models were reviewed by the community working group and other experts in the ecology of the Kimberley region and revised accordingly (Sarah Legge, pers. comm.). For the remaining 10 animals and all the rare plants, habitat and distribution are not well-defined in literature or reliably predicted with existing spatial datasets, so we judged these species “data deficient” and did not develop habitat models.

2.5. Comparing spatial patterns of cultural/social values with biodiversity values

To assess the relationships between cultural/social and biodiversity targets, we summarized the thematic associations and spatial relationships between cultural/heritage site attributes, threatened species habitat, and landscape features, and specifically freshwater features. To illustrate distribution patterns of cultural/heritage sites and threatened species habitat across the study area, we created a grid of $3 \times 3$ km cells and sampled the count of cultural/heritage sites per cell and the count of threatened species with modeled habitat occurring in each cell (Fig. 3).

2.6. Landscape measures of access and disturbance

The availability and provision of native game animals and bush tucker/medicine plants, and any ecosystem service, requires consideration of two components: supply of ecosystem services, and
physical and legal access to the services (Tallis et al., 2015). To measure and map the pattern of relative accessibility across the study area, we calculated a spatial metric of access as the sum of proximity to Nyikina Mangala communities and the proximity to roads (Fig. 4), with proximity measured as the inverse of euclidean distance from each population center and road segment to the edge of the NTD. The result is a measure of ecosystem service provision in terms of access for any part of the landscape and any feature. Data sources and calculations are documented in Appendix 3.

Similarly, the abundance and viability of native game animals and bush tucker/medicine plants, and the provision of other ecosystem services, depends on current ecological condition and historic disturbances (Woinarski et al., 2007; Raiter et al., 2014). To estimate and map patterns of ecological disturbance, we developed two spatial measures. The first is a spatial index of disturbance from infrastructure and human land use (Fig. 5) derived from available public spatial datasets representing population centers, roads, mine operations, petroleum operations, local hydrological alteration (dam walls, canals), livestock use (bores, water pumps, tanks), and other infrastructure (airports, power lines, fences). Data sources and calculations are documented in Appendix 3. The result is a coarse, generalized measure of cumulative impacts. The second metric is the frequency of destructive late-season fires between 2000 and 2015 recorded by NAFI (2016), shown in Fig. 6. Late dry season fires occur after July 31, burning hotter and over larger extents than in the early dry season, and are ecologically destructive and an urgent threat to biodiversity in the region (Woinarski et al., 2011; Carwardine et al., 2012; Bartolo et al., 2012). Fires are monitored and recorded in public datasets by NAFI.

2.7. Decision framework for mitigation

Through a series of workshops, the community working group developed a framework to assess development proposals and define conditions for negotiation of mitigation measures according to the types of spatial targets affected and the accessibility and ecological condition of these targets (see Fig. 7). The framework follows steps in the mitigation hierarchy to avoid, minimize, and offset impacts.

To enable the Nyikina Mangala community to conduct rapid spatial analysis of the potential impacts of development proposals, we developed a Geographic Information System (GIS) software application that measures and reports the types and amounts of targets occurring in a user-defined proposed development footprint or impact area. The

Fig. 3. Spatial pattern of aggregated social/cultural targets and biodiversity targets.
application also analyses and reports the area of the footprint that lies in each of the three classes of access, three classes of disturbance from the cumulative impacts of land use and infrastructure, and four classes of destructive fire frequency.

3. Results

Based on existing survey of cultural/heritage sites, 41% of sites are thematically linked to freshwater based on the site attributes. Almost 70% of sites occur within a kilometer of a water body or the floodplains and riverine wetlands of the Fitzroy River and major tributaries. Cultural/heritage sites are also more abundant near rocky hills and outcrops.

Of the 22 threatened species for which we developed spatial habitat models, potential habitat of 17 or 77% of modeled species occurs in the Fitzroy River floodplain and riverine wetlands, and for 13 or 60% of those species, potential habitat occurs exclusively in the Fitzroy River floodplain. All 19 migratory shorebirds protected by international agreements also use the Fitzroy River floodplains and riverine wetlands seasonally during the wet season. Potential habitat of four modeled species includes rocky hills and outcrops – Black-flanked Rock-wallabies use rocky hills exclusively, while Northern Quoll and two threatened bat species use rocky hills as refuge habitat and for denning and roosting.

Fig. 3 shows the general distribution of surveyed cultural/heritage sites and potential habitat of protected species in relation to the Fitzroy River floodplain. To protect the specific locations of cultural/heritage sites, the map spans only a 60 × 90 km portion of the NTD and the datasets are resampled in a 3 km resolution grid. Cultural/heritage sites have not been completely or consistently surveyed across the NTD, so gaps and low values are likely areas that have not been surveyed or for which survey data was not available. Social/cultural targets also include native game animals and bush tucker/medicine plants that are present across the landscape but are not quantified in terms of abundance.

The mitigation framework (Fig. 7) defines conditions for negotiation of mitigation measures following steps in the mitigation hierarchy to avoid, minimize, and offset impacts. The community working group defined avoidance areas for developments in the NTD as 1) cultural/heritage sites including a two kilometer buffer zone around each site and 2) freshwater protection zones defined and mapped as the floodplains and riverine wetlands of the Lower Fitzroy River, the Fraser Rivers, and their major tributaries inside the NTD. The defined avoidance areas for cultural/heritage sites and freshwater features cover approximately 13% and 12% of the NTD, respectively. Together, the two protection zones cover 21% of the NTD. The landscape measures of access (Fig. 4) and ecological condition (Figs. 5 and 6) provide measures of ecosystem services provision and inform steps to minimize and offset impacts.

4. Discussion

Environmental impact assessments (EIAs) are intended to minimize risks to environmental values and human rights, lessen adverse impacts, and strengthen positive outcomes of business investments. For an EIA to fulfill this purpose, it must consider the perspectives of everyone affected by a developer’s operations. Too often, developers ignore social and cultural impacts, focusing instead on environmental assets that often do not fully represent a community’s values, and in doing so, forfeit the opportunity to minimize human rights violations and costly
conflicts. Here we present a practical framework and process that can be applied proactively to assess impacts to environmental, social and cultural values. We discuss application of this proactive planning approach to the Nyikina Mangala Native Title Determination (NTD) in Northern Australia as well as technical capacity needed to expand implementation more broadly.

In the Nyikina Mangala NTD, there is a strong thematic and spatial relationship between cultural/heritage sites and freshwater features, and the Lower Fitzroy River in particular. Biodiversity, represented by potential habitat for threatened animals, is also concentrated in Lower Fitzroy freshwater systems. Both cultural/heritage sites and threatened species habitat also show a strong spatial relationship in rocky hills and outcrops. A significant fraction of cultural/heritage sites are located near rocky hills, and four threatened species use rocky hills, one (Black-footed Rock-wallaby) exclusively.

The concentration of social/cultural and biodiversity values around freshwater features may be expected in arid climates where human settlements, species richness, and ecosystem productivity are highly dependent on water availability (e.g. Davis et al., 2017). The Fitzroy River and its tributaries provide multiple ecosystem services including water, game animals, bush tucker/medicine plants, and habitat for threatened species. Similarly, rocky hills have value for historic human settlements and as unique habitat for native plants and animals (e.g. Fitzsimons and Michael, 2017). However, cultural/heritage sites were not surveyed systematically across the NTD, and there is likely some survey bias for areas near the Fitzroy River and rocky hills due to higher access.

Though the Fitzroy River provides critical social/cultural values and biodiversity values, much of the riparian zone, riverine wetlands and water bodies have been degraded by livestock grazing (Morgan et al., 2004; Watson et al., 2011), and fish passage and freshwater habitat connectivity have been impaired by the Camballin barrage (Morgan et al., 2005). The river is also threatened by future development (Australian Government, 2015; Department of Primary Industries and Regional Development, 2017; Morán-Ordóñez et al., 2017). Water quality and flows are affected by withdrawals, sedimentation, and pollution across the watershed. Although not the focus of the current study, any impact assessment of development projects in the watersheds of the Fitzroy River and Fraser Rivers, including projects in the upper basins outside the NTD, should evaluate impacts to water quality and quantity in the downstream sections of the river inside the NTD.

The decision framework developed here is a means to ensure FPIC is possible for communities within existing mechanisms, and allow communities to shift from a reactive role to a pro-active role in development processes. We mapped targets defined in the Healthy Country Plan: cultural/heritage sites, freshwater features, common native animals and plants represented by biophysical habitat types, and legally-protected threatened and migratory species represented by potential habitat models. The community defined protection zones for cultural/heritage sites and freshwater features that cover 21% of the NTD. To represent differences in provision and viability of native animals and plants and other ecosystem services, we developed spatial measures of access and ecological condition.

This spatial information can be the basis to proactively apply the mitigation hierarchy – first avoid, then minimize, and if appropriate also offset impacts – to balance conservation objectives with impacts...
associated with future potential development (see Fig. 7). The high priority conservation areas identified to avoid development impacts to cultural/heritage sites and freshwater features cover approximately 21% of the NTD. Though the cultural/heritage sites dataset is incomplete and the avoidance area will likely expand, the 21% figure suggests that some conflicts could potentially be resolved by redesigning development footprints to avoid impacts to those conservation targets. Mitigation recommendations can be defined based on the location and the nature and distribution of conservation targets affected. Where proposed development overlaps highly irreplaceable targets, greater emphasis should be given to avoidance than minimization. In some areas and for some targets, offsets may be appropriate to further mitigate impacts.

Biodiversity offsets within the Mitigation Hierarchy have been used by all Australian states and territories, and by the Australian Government where a development is likely to impact on matters of national environmental significance under the Environment Protection and Biodiversity Conservation Act 1999 (Fitzsimons et al., 2014; Hawdon et al., 2015; Maron et al., 2015). These schemes vary by jurisdiction, in the types of biodiversity matters considered, in the metrics used to assess impact and determine offsets, and instruments and guidance used to implement them (e.g. DSEWPC, 2012). Nonetheless, they typically consider ecological communities (typically vegetation types) or threatened species (and their habitats).

Areas that are more accessible or that support intact habitat in good ecological condition may necessitate a higher requirement for mitigation of impacts from development projects and other land use changes (McKenney and Kiesecker, 2010; Villarroya et al., 2014). Accessibility and ecological condition, as represented by the access and disturbance measures, indicate greater provision ecosystem services or abundance of native plants and animals including rare and threatened species. These measures can inform decisions about the conservation significance and mitigation burden of development in any given location (see examples in Fig. 7).

Packaging cultural and social data at a landscape scale can also guide other management decisions in the NTD. Sites that occur in highly accessible and/or highly disturbed areas could benefit from management plans and actions such as fences, walkways, and signage to reduce risk of degradation. Disturbance measures may also guide restoration and threat management actions such as fire and grazing management and invasive species control. Management actions for biodiversity in the Kimberley region have been studied and prioritized by Carwardine et al. (2012) in terms of cost effectiveness.

Australia was one of the first countries to require free, prior and informed consent in local legislation (MacKay, 2004). Considering the stated plans of national and state governments to further develop northern Australia, there is a timely opportunity to enhance current development assessment processes to better incorporate Indigenous social/cultural values, as outlined in this paper. Considering the significant area in Northern Australia to which Indigenous people have Native Title and rights to FPIC, incorporating such processes would improve the social, cultural and environmental outcomes of development proposals and reduce conflicts.

Some legislative and policy instruments already in place will benefit from proactive planning. For example, Native Title holders have the right to negotiate development proposals that impact their native title rights and interests – which also leads to rights to compensation if there are subsequent impacts on native title rights and interests. Improved
quality of information and analysis will contribute to more informed negotiation and improve implementation of cultural heritage protection requirements at both Federal and State government levels (e.g. Federal Aboriginal and Torres Strait Islander Heritage Protection Act 1984; Western Australia Aboriginal Heritage Act 1972). Native Title Representative bodies (NTRB) that hold a statutory role to represent groups of Registered Native Title Body Corporates (RNTBCs) and the RNTBCs themselves are faced with such a high volume of exploration license applications and other development proposals that reviewing and responding to each proposal is nearly impossible due to limited capacity and time. A spatial framework similar to what we have developed that allows identification of areas with high values and high vulnerability would enable NTRBs and RNTBCs to prioritize and focus limited resources on high-risk or high-conflict proposals.

For development projects that are likely to impact biodiversity values such as threatened species and ecological communities, Federal (EPBC 1999) and state or territory (Western Australia Biodiversity Conservation Act 2016) legislation require impact assessments prior to permit application. These assessments are typically made by consulting companies and use existing public datasets, but may collect new biodiversity data depending on the size of the project and the likelihood of impact to a highly threatened species. Governments will then assess the suitability of the proposed development and approve, request modifications, or reject the proposal, depending on the type and range of species affected. This may vary by state/territory jurisdiction. The threatened species potential habitat models developed for this study indicate what legally-protected species might occur in or be affected by a proposed development site, for internal reference by the community, and may inform surveys conducted as part of the impact assessment process.

### 4.1. Data use, limitations and sensitivities

Proactive planning can benefit both traditional owners and developers. For traditional owners, planning and organization is critical to FPIC, enabling timely decisions about avoidance and mitigation and strengthening negotiating position. These proactive decisions can also steer investments away from areas of conflict, saving time and expense for all parties. However, spatial planning requires spatial data, which is often incomplete. In particular, the coverage of cultural/heritage sites and threatened species records depends on survey effort, and areas without survey records may contain un-recorded sites and species. In this study, the cultural/heritage sites dataset was compiled from 18 different sources with varying survey designs and extents, leaving large portions of the NTD where data was not available. Because the cultural/heritage sites survey reports do not include absence data, it’s impossible to estimate or distinguish unsurveyed areas from areas without sites. Local surveys for cultural/heritage features and threatened species are a critical part of EIA in the exploration phase of any development project, but are limited in extent to each development site. This underscores the need for proactive, landscape-level surveys. Funding for regional survey efforts will be a critical limiting factor if landscape-level proactive planning is to be conducted more widely. A useful precedent for funding proactive regional planning is Healthy Country Planning in Australia that began with several workshop and pilot studies supported by The Nature Conservancy that developed a replicable model and demonstrated its utility. Since then, Healthy Country Planning has been applied in over 140 projects by more than 20 organizations with funds...
from various sources including Aboriginal corporations, NGOs, government, foundations, and the private sector (Carr et al., 2017). Technical capacity for collecting and managing survey data has improved across Australia with GPS survey software such as Fulcrum (Spatial Networks, Inc., 2018) and CyberTracker (Ansell and Koenig, 2011) and with online spatial information platforms such as the Atlas of Living Australia (2018), Northern Australia Fire Information (2016), Queensland Globe (Queensland DNRE, 2018), and Western Australia Landgate (Western Australian Land Information Authority, 2018).

The process of compiling general predictive models to map conservation targets can guide survey efforts. Like many parts of the world, the Nyikina Mangala NTD and the Kimberley region lack comprehensive surveys and datasets describing the distribution of native animals and plants, from relatively common game species and bush tucker/medicine plants to rare and threatened biodiversity (McKenzie et al., 2009; Carwardine et al., 2011). The biophysical habitat classification and the disturbance index created as part of this assessment may guide surveys for both site-level impact assessments in the short term and landscape-level sampling designs across the NTD.

Bringing sensitive and threatened features into spatial planning while protecting their locations presents a challenge. For this study, the Nyikina Mangala community compiled a detailed dataset of cultural/heritage sites for internal use and allowed the broad summary of their cultural data for external stakeholders, but have chosen to keep the precise locations private to preserve and protect these values, as there is evidence that publishing locations to aid planning and conservation could harm the same values (Lindenmayer and Scheele, 2017). However, there is already precedent in the fields of paleontology and archaeology that advance restrictions on the publication of site locations and the promotion of government policies and regulations to limit collection and trade in artefacts and culturally sensitive important material. There is also precedent in Australia where the High Court can hear cultural stories in closed sessions in determining connection to country for Native Title determinations. Indigenous communities and aboriginal corporations must have confidence that secure mechanisms are in place for sharing sensitive spatial information to proactively inform and guide development plans while protecting locations. This will require new tools and approaches to data sensitivity and access.

To enable the Nyikina Mangala community to conduct rapid assessments of the potential impacts of development proposals in the NTD, we developed a Geographic Information System (GIS) software application that measures and reports the types and amounts of targets occurring in a user-defined proposed impact area. This allows the Walalakoo Aboriginal Corporation to facilitate community decision-making by reporting and comparing various development scenarios. A capacity-building program is underway that includes application testing, GIS software training, and development of a technical user manual.

Cultural assessments face other methodological challenges in addition to limited availability of comprehensive and current spatial data. Not all cultural values are readily mapped or measured spatially. Intangible values that cannot be mapped such as spiritual beliefs, language, and oral history are necessary to maintain culture (Partial and Dunphy, 2016; Watson et al., 2011). Also, cultural values are not static and will change over time. Threatened species listings will also change over time, as many northern Australian mammal populations are experiencing a decline (Fitzsimons et al., 2010), and many of these species have not yet been listed under state/national threatened species legislation. Therefore, planning frameworks like this must be adaptive and allow for regular updates and revision.

4.2. Future directions/conclusions

There is an urgent need to transform development planning from reactive site-level planning for individual projects to consider landscape-level development scenarios in advance of proposed development projects (Kiesecker and Naugle, 2017; Kiesecker et al., 2017). In view of the FPIC principles, all development projects affecting the lives of Indigenous peoples require their early and sustained input to ensure that projects mitigate impacts to social and cultural values and reflect their choices of development (UN, 2008). With this case study we illustrate that proactively compiling social and cultural values is possible and practical. This can strengthen traditional Indigenous governance systems, reinforcing the role of Indigenous peoples in the decision-making process and improving their position to negotiate with other parties, be they local or national authorities, the private sector, or international development institutions.

First and foremost, Indigenous peoples need an opportunity to strengthen their individual and collective capabilities to exercise their rights and have a greater say in decisions that affect their values and futures. Healthy Country Planning (Carr et al., 2017) can provide a clear articulation of community values and objectives for management of their own land. This provides a foundation for defining and mapping targets in a spatial decision-making framework and analyzing these targets in existing legal and policy contexts, including threatened species and cultural heritage legislation. Spatial planning requires training and capacity building in both the technical aspects of spatial planning and in the effective analysis and interpretation of results is required. Additionally, the effective use of spatial planning for decision-making requires capacity for analysis of results in the context of the relevant legal and policy environment.

The fields of conservation planning and mitigation planning for biodiversity have produced best practices and data systems to help facilitate effective impact assessment. These include criteria for prioritizing protection of species and habitat areas based on concepts of rarity and vulnerability (Tallis et al., 2015) and spatial frameworks that identify conflicts between development proposals and with conservation goals (Saenz et al., 2013). These have been widely adopted in EIA law and policy (Villarroya et al., 2014) and are recognized by developers and lenders (IFC, 2012), with resulting benefits for biodiversity conservation. Similar criteria and frameworks for social and cultural values have not been universally accepted. As Indigenous communities define these criteria, this will help facilitate and strengthen the incorporation of their values into development approval processes.

Given growing global resource demands (Oakleaf et al., 2015), land use conflicts are likely to increase with profound implications for both biodiversity and Indigenous land values. Incorporating the likelihood of future change into land-use planning can alleviate uncertainty and ultimately make societal adaptation to change more efficient and less costly (Kennedy et al., 2016a, 2016b). Predicting and quantifying future impacts can help to justify proactive protection of places important to Indigenous communities and biodiversity and to underscore the consequences of failing to do so (Kiesecker et al., 2017). We hope our study will motivate regulatory agencies and land managers to proactively map social, cultural, and biodiversity values and forecast impacts at the landscape level, and use this information to avoid a business-as-usual development trajectory. Proactive planning to predict and avoid impacts to social and biological values will, in the long run, be the less costly and more sustainable path.

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Declarations of interest

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2018.09.002.

References


Conservation status of the Oyster Reef Ecosystem of Southern and Eastern Australia

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\textbf{Abstract}

Reef ecosystems all over the world are in decline and managers urgently need information that can assess management interventions and set national conservation targets. We assess the conservation status and risk of ecosystem collapse for the Oyster Reef Ecosystem of Southern and Eastern Australia, which comprises two community sub-types established by 	extit{Saccostrea glomerata} (Sydney rock oyster) and 	extit{Ostrea angasi} (Australian flat oyster), consistent with the IUCN Red List of Ecosystems risk assessment process. We established: (i) key aspects of the ecosystem including: ecological description, biological characteristics, condition and collapse thresholds, natural and threatening processes; (ii) previous and current extent of occurrence and current area of occupancy; and (iii) its likelihood of collapse within the next 50–100 years. The most severe risk rating occurred for Criterion A: Reduction in Extent (since 1750) and Criterion D: Disruption of biotic processes (since 1750), although assessment varied from Least Concern to Critically Endangered amongst the four criteria assessed. Our overall assessment ranks the risk of collapse for the ecosystem (including both community sub-types) as Critically Endangered with a high degree of confidence. Our results suggest the need for rapid intervention to protect remaining reefs and undertake restoration at suitable sites. Several restoration projects have already demonstrated this is feasible, and Australia is well equipped with government policies and regulatory mechanisms to support the future conservation and recovery of temperate oyster ecosystems.

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2351-9894/© 2020 The Authors. Published by Elsevier B.V. This is an open access article under the CC BY-NC-ND license (http://creativecommons.org/licenses/by-nc-nd/4.0/).
1. Introduction

Shellfish reef ecosystems develop when high densities of shellfish, typically oysters or mussels, occur and form biogenic structures that function as ecosystem engineers and the foundation of the ecosystem. Shellfish reef ecosystems support important environmental characteristics, such as unique assemblages of associated fauna and valuable ecosystem services, including fish production, coastal protection, erosion mitigation, pH buffering and nutrient cycling (Coen et al., 2007). These services have been valued at between US$5,500 and $99,000 ha⁻¹ (2011 dollars; Grabowski et al., 2012). Because of these valuable services, the protection and restoration of shellfish ecosystems are of interest to coastal managers as one potential natural solution to ameliorating the impacts of climate change, coastal eutrophication and habitat degradation (zu Ermgassen et al., 2016; Cohen-Shacham et al., 2019; McLeod et al., 2019).

Shellfish reefs are globally distributed occupying intertidal and shallow subtidal zones in estuaries and on open coastlines across temperate and tropical environments. Today, however, over 85% of oyster reef ecosystems globally have been lost or degraded (Beck et al., 2011). Mechanisms for losses include: overharvest of shellfish and reef degradation from physical removal or breaking up reefs during harvest, changes in abiotic conditions such as salinity, sedimentation, hypoxia and flow due to upper catchment and shoreline modification, disease and pollution (Holmes, 1927; Kirby, 2004; Beck et al., 2011; Gillies et al., 2018; Pogoda, 2019). Consequently, oyster reef ecosystems are considered one of the most imperiled and threatened marine ecosystems globally (Beck et al., 2011). Although their decline and associated need for conservation is increasingly recognised as a priority amongst conservation groups and professional science networks (e.g. Beck et al., 2011; zu Ermgassen et al., 2016; Fitzsimons et al., 2019; Pogoda et al., 2019; https://www.shellfishrestoration.org.au) there is a ubiquitous absence of protection (be it legal or policy) or global recognition of the threat of ecosystem collapse.

In Australia, oyster reef ecosystems can be formed by at least 14 different oyster and mussel species and occur in both tropical and temperate regions (Gillies et al., 2018). Of these species, there is reliable evidence that reefs formed primarily by Ostrea angasi (Australian flat oyster) or Saccostrea glomerata (Sydney rock oyster) have undergone considerable decline from historical distributions (Kirby, 2004; Alleway and Connell, 2015; Gillies et al., 2015a, 2018; Ford and Hamer, 2016). Here we provide a description of the ecosystem these species form and complete a risk assessment using the IUCN Red List of Ecosystems framework (https://iucnrs.org/sub-types) to assess the status of shellfish reef ecosystems formed by O. angasi and S. glomerata oysters. We term the ecosystem the ‘Oyster Reef Ecosystem of Southern and Eastern Australia’ (SEA Oyster Reefs) which comprises of two community sub-type developed by the above species. We assess the entire ecosystem and, where possible, provide specific information for each community sub-type. The risk assessment considers five criteria, each with three sub-criteria, to define numerical thresholds of threat from Least Concern (LC) through to Critically Endangered (CR) (Rodriguez et al., 2015). The approach is consistent with assessments made according to the IUCN Red List of Species and is similar to the assessment process for ecosystems under the Australian Commonwealth Government’s environmental protection legislation (Environment Protection and Biodiversity Conservation Act 1999). Assessing the risk of collapse of ecosystems provides vital understanding about the root causes of decline and potential methods for recovery and can inform appropriate governmental and intergovernmental protection levels and mechanisms (Rodriguez et al., 2015).

1.1. Ecosystem description and key biological characteristics

No single global definition of an ‘oyster reef ecosystem’ exists, largely because reef systems differ considerably according to their foundational species, location, surrounding abiotic attributes and biological processes. Kasoar et al. (2015, p. 982) provide the most quantitative and adaptable definition relevant to Australian oyster reef ecosystems: “Bivalve reef [ecosystems] consists of large areas of biogenic habitat, dominated by living bivalves where the complex structure of hard shells supports a distinct community that is persistent through time”. Kasoar et al. (2015, p. 982) then expand on this general definition: “‘large areas’ typically consist of multiple patches, at least some of which are larger than 5 m²; ‘dominated’ means at least 25% cover of live shell matter across that space — non-living shell (culch) may further add to habitat structure and to continuity over time, but without new growth they are unlikely to persist; a ‘distinct community’ is one that supports species and interactions that are rare or absent in surrounding communities; and ‘persistent through time’ describes communities that are likely to remain over ‘decadal time scales or longer’”.

Both S. glomerata and O. angasi provide the physical and biogenic structure and exhibit similar physical forms and biological composition. Structure occurs as either low-profile beds or high-profile reefs, which are developed through clustering of oysters, on soft sediments or hard structures, in high density. These species also support a similar community assemblage consisting of the same or similar functional species of mobile fauna, epifauna, fishes and microbiota (Crawford et al., 2020; McLeod et al., 2020). Because both species provide a similar physical and biogenic structure and similar or identical ecosystem services in coastal environments (Fig. 1) and are the most common reef-forming species in southern and eastern Australia, we considered these as two distinct community sub-types of a single ecosystem.

Based on historical and current observations collected on both community sub-types (Ford and Hamer, 2016; Gillies et al., 2017; McAlee et al., 2016; Keane and Gardner, 2018; Crawford et al., 2020; McLeod et al., 2020) we provide a qualitative description of the physical form and functional features of SEA Oyster Reefs at the patch-scale (i.e. network of reefs within an estuary, its most typical form of occurrence) to aid the delineation of reefs ecosystems compared to other ecosystems (i.e. oyster reefs versus dense populations of oysters within other ecosystems) (Table 1).
1.2. Abiotic environment and distribution

The abiotic envelope in which SEA Oyster Reefs ranges from estuarine to full marine waters in moderate to low energy environments (Edgar, 1998; Gillies et al., 2015a). The ecosystem occupies the intertidal and subtidal zone between the mean high tide line to 30 m below sea level, in estuaries, bays, inlets, gulfs and coastal waters from southwestern Western Australia, eastward along the southern coast, including Tasmania, to south-east Queensland south of Bundaberg (Gillies et al., 2018). Oyster reef ecosystems formed by *O. angasi* typically occur subtidally, from low intertidal to a depth of 30 m and favour fully marine salinities. *S. glomerata* typically occur in the intertidal zone within estuaries although historic evidence suggests reefs were common in subtidal areas down to at least 10 m (Smith, 1981; Diggles, 2013) and prefer more estuarine salinities (10–35 ppt) (Dove and O’Connor, 2007). Community assembles are functionally similar amongst both community sub-types with some overlap in species composition where ranges overlap (Crawford et al., 2020; McLeod et al., 2020).

1.3. Current typological classifications

Shellfish Beds and Reefs are classified under the global IUCN Global Ecosystem Typology (Keith et al., 2020) as ecosystems occurring within the Marine Realm, Marine Shelves Biome (M1.4). In Australia, oyster reef ecosystems can be classified under the National Intertidal/Subtidal Benthic (NISB) habitat classification scheme (Mount et al., 2007) and Interim Australian National Aquatic Ecosystem Classification Framework (Aquatic Ecosystems Task Group, 2012) as occurring in marine and estuary systems on unconsolidated substrate with a Structural Macrobiota (SMB) dominated by a filter feeding assemblage. The ecosystem is classified under the Ramsar Classification System for Wetland Type (Ramsar, 2012) and is defined as E7 ‘Bivalve (shellfish) reefs’.

1.4. Key natural processes

Oyster reefs typically form as successive generations of bivalves settle and grow on top of one another and persist by several key processes and interactions (Fig. 2). The availability of clean substrate is a key requirement for regular recruitment and reef persistence, with oysters showing a preference for attaching to other living oysters (Rodriguez-Perez et al., 2019). This process aids the physical development of reefs and creation of a positive shell budget where new oysters settle onto live or dead oysters, elevating the reef from the surrounding substrate. A high spawning biomass, where survival of settled larvae through to maturity is greater than adult mortality is required to support reef growth and maintain dense aggregations (Powers et al., 2009).

The location of reefs and beds within an area can shift through time (across decadal time scales) and geological and Aboriginal cultural evidence of food middens indicates the potential of populations to persist for very long (at least centennial) time periods in a single location (Edgar and Samson, 2004; Gillies et al., 2015a). A combination of environmental parameters govern the position of oyster reef ecosystems within a seascape, including: wave exposure and currents, sedimentation, salinity, food availability and suitability of substrate for settlement. Both oyster species are subject to diseases, which are known to inflict significant mortality in aquaculture settings (Winter Mortality Syndrome, Queensland Unknown Disease (QX) for *S. glomerata; Bonamia exitosa* for *O. angasi*) (Nell, 2001; Carnegie et al., 2014).

A key feature and biological process of oyster reef ecosystems is their capacity to capture food and nutrients from the water column and transfer them to the benthos, a process known as benthic-pelagic coupling (Newell, 2004). The drawdown of plankton and seston from the water column through the filter feeding of oysters and the subsequent production of oyster biomass, faeces and pseudo faeces, cleans the water-column and enriches the benthos with nutrients that underpin the...
productive of benthic fauna and vegetative communities (Dame et al., 1984; Newell and Koch, 2004), while facilitating microbial activity that positively influence nitrogen and phosphate re-mineralization (Kellogg et al., 2013).

1.5. Historical and current threatening processes

Threats to SEA Oyster Reefs mirror global patterns (Beck et al., 2011). Historical threats were primarily unregulated fishing resulting in over-harvest during the first 100 years of European colonisation and the use of destructive fishing equipment such as dredges (Smith, 1981; Nell, 2001). Oyster fishers used dredges, but also hand harvest methods, which broke up, removed or buried oysters and shell resulting in loss of oyster biomass, removal of settlement substrate, a decline in ecosystem function, and ultimately a shift towards an unconsolidated substrate. Abiotic factors such as historical and ongoing changes to land and water use in catchments and estuaries can threaten ecosystem formation and persistence by influencing the environmental conditions of an estuary (e.g. salinity, pH, dissolved oxygen, freshwater flow, tidal dynamics, sedimentation, shoreline availability, auto and allochthonous estuary primary production (Chan et al., 2002; Taylor et al., 2004; Thrush et al., 2004)). These drivers have a direct impact on oyster growth and survival by controlling the degree of smothering, water quality, availability of surface for recruitment, food availability, and predation (Lenihan and Peterson, 1998; Nell, 2001; Brumbaugh et al., 2006; Wasson, 2010; Diggles, 2013; O’Connor et al., 2015). The oxidation of sulfidic floodplain sediments and release of acidic waters (pH < 6) into estuaries is particularly widespread in eastern Australia (Sammut et al., 1996) and causes significant mortality and stress in S. glomerata (Dove and Sammut, 2007), although oysters may be adapting (Amaral et al., 2011). Floods in historical and contemporary times are catastrophic threats, which can cause physical damage, abiotic changes in estuaries and precipitate the spread of diseases (e.g. QX, Winter Mortality) and parasites such as mudworm (Ogburn et al., 2007; Green et al., 2011; Diggles, 2013; Spiers et al., 2014). Current threats, in addition to the legacy of historical harvesting and catchment disturbance, include disease (described above), climate change (primarily through ocean acidification), altered temperature and salinity and resultant potential loss of suitable abiotic growing conditions (Parker et al., 2009; Gillanders et al., 2011), commercial and recreational fishing (Keane and Gardner, 2018), and removal of available surfaces for colonisation through shoreline modification.

1.6. Definition of ecosystem collapse

Whilst the IUCN Red List of Ecosystems provides a mechanism to assess ecosystem collapse across the extent of the entire ecosystem, we were unable to find a definition of degradation towards collapse at the local level (i.e. at a location — the scale at which most management is undertaken) in the literature for any shellfish reef ecosystem. We therefore provide a definition of ecosystem collapse for a reef system at the location scale derived from our cause-effect model (Fig. 2), from the Interim Australian National Aquatic Ecosystem Classification Framework (Aquatic Ecosystems Task Group, 2012) and common criteria used to measure the success of oyster reef restoration in the United States and Australia (Oyster Metrics Workgroup, 2011; Baggett et al., 2014; Gillies et al., 2017; McLeod et al., 2020).

Table 1
Semi-qualitative reef attributes (physical form and functional features) of the Oyster Reef Ecosystem of Southern and Eastern Australia which may aid the delineation of reefs ecosystems versus alternate ecosystems with oyster populations.

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Fully functional reef ecosystems</th>
<th>Partially functional reef ecosystems</th>
<th>Oyster populations within alternate ecosystems</th>
<th>Oyster density (μ m² ± s.d.) and sources</th>
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<tbody>
<tr>
<td>1. Oyster density</td>
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<tr>
<td>O. angasi</td>
<td>&gt;50 live oysters/m²</td>
<td>50-10 live oysters/m²</td>
<td>&lt;10 live oysters/m²</td>
<td>Jones and Gardner (2016) 18.3 ± 16.7</td>
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<td>Crawford et al. (2020) 20 ± 1 to 229 ± 7</td>
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<td>Summerhayes et al. (2009)</td>
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<td>940 ± 251</td>
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<td>McLeod et al. (2020) 10.2 ± 3.3 to</td>
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<td></td>
<td>740.5 ± 15.8</td>
</tr>
<tr>
<td>S. glomerata</td>
<td>&gt;500 live oysters/m²</td>
<td>500-100 live oysters/m²</td>
<td>&lt;100 live oysters/m²</td>
<td>Powers et al. (2009)</td>
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<td>Schultz and Burke (2014)</td>
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<tr>
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<td></td>
<td></td>
<td></td>
<td>Powell et al. (2006)</td>
</tr>
<tr>
<td>2. Oyster coverage/dominance and height</td>
<td>Oysters and oyster shell are the primary physical feature in seascape</td>
<td>Oysters and oyster shell partially cover seascape, interspersed with other physical, biological features</td>
<td>Oysters and oyster shells minor feature in seascape</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3. Shell budget and height</td>
<td>Increasing or stable spatial extent and/or height. Patches consist of a mix of live oysters and dead shell.</td>
<td>Little or no evidence of stable shell structure</td>
<td>Few or no discrete oyster reef/shell patches</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>4. Patch number and size</td>
<td>Multiple patches of reef with vertical relief from surrounding substrate, reef patch sizes ≥ 5m²</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

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The ecosystem has collapsed when there are no remaining locations dominated by living oysters and oyster shells. Spatial complexity and the presence of hard substrate will have significantly decreased (where not occurring on otherwise hard surfaces (e.g. rock or mangrove roots). Microclimates and local hydrodynamics may also change. Species assemblages will shift from a diverse range of sessile and mobile reef-associated organisms, to a system that is predominantly characterised by infauna and deposit feeders (when shifting to soft sediments) or lower diversity and biomass of reef-associated species when shifting to bare rock/mangrove). Indicators of ecosystem decline at the patch-scale can be observed by measuring density of oysters, oyster recruitment, survival and growth (Table 1).

2. Risk assessment methods

Following the methods of Keith et al. (2013) and guidelines of Rodriguez et al. (2015), we conducted a risk assessment to determine the risk of collapse for SEA Oyster Reefs comprising the community sub-types S. glomerata and O. angasi. Five criteria and three sub-criteria, developed for the IUCN’s Red List of Ecosystems (Rodriguez et al., 2015; https://iucnrle.org/), formed the framework of the risk assessment. These were: Criterion A) rates of decline in ecosystem distribution; Criterion B) restricted distributions with continuing declines or threats; Criterion C) rates of environmental (abiotic) degradation; Criterion D) rates of disruption to biotic processes; and Criterion E) quantitative estimates of the risk of ecosystem collapse. The sub-criteria in each primary criteria define timeframes for the assessment period (e.g. past 50 years, next 50 years and since 1750), over which decline (or degradation) in ecosystem extent (or function) can be assessed (see Table 2 for all criteria and sub-criteria). Metrics, defined in Keith et al. (2013), were used to assign one of six risk categories to the ecosystem for each sub-criterion and included: data deficient (DD), least concern (LC), near threatened (NT), vulnerable (VU), endangered (EN) and critically endangered (CR).

Table 2
Assessment of threat ranking of the Oyster Reef Ecosystem of Southern and Eastern Australia using the IUCN Red List of Ecosystems criterion.

<table>
<thead>
<tr>
<th>Sub-criterion</th>
<th>Criterion A: Reduction in extent</th>
<th>Criterion B: Restricted geographic distribution</th>
<th>Criterion C: Environmental degradation</th>
<th>Criterion D: Disruption of biotic processes</th>
<th>Criterion E: Quantitative analysis</th>
<th>Overall threat ranking (based on highest risk ranking)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Past 50 yrs</td>
<td>DD</td>
<td>LC</td>
<td>DD</td>
<td>DD</td>
<td>1 ≤ 50% in 50 yrs</td>
<td>DD</td>
</tr>
<tr>
<td>2 Next 50 yrs</td>
<td>DD</td>
<td>EN</td>
<td>DD</td>
<td>DD</td>
<td>2 ≤ 20% in 50 yrs</td>
<td>DD</td>
</tr>
<tr>
<td>3 Since 1750</td>
<td>CR</td>
<td>VU</td>
<td>CR</td>
<td>DD</td>
<td>3 ≤ 10% within 100 yrs</td>
<td>CR</td>
</tr>
</tbody>
</table>

DD = Data Deficient, LC = Least Concern, NT = Near Threatened, VU = Vulnerable, EN = Endangered, CR = Critically Endangered.
2.1. Sources of data and assessment methods

2.1.1. Criterion A: Reduction in geographic distribution

For Criterion A, we used the published literature sources of Gillies et al. (2015a, 2018), which provide data at the national scale and several other studies which described historical distributions at regional scales (i.e. state jurisdiction: Kirby, 2004; Ogburn et al., 2007; Diggles, 2013; Alleway and Connell, 2015; Ford and Hamer, 2016; Thurstan et al., 2020) as a proxy for ecosystem distribution, since no current or previous distribution maps exist. We assessed historical distributions at the location level with knowledge of present distributions published in Gillies et al. (2015a), Jones and Gardner (2016) and McLeod et al. (2020).

Information from the above studies consisted of a mix of primary and secondary sources that include: early explorer accounts, fisheries and government reports, commercial fishery surveys, first person accounts (published in newspaper articles), archaeological excavations (aboriginal middens), sediment cores, place names and reviews of fisheries legislation. Most of the scientific studies described the ecosystem in the context of wild oyster fisheries/oyster harvest and used a combination of fisheries harvest records, cultural histories, eyewitness accounts and parliamentary records as attesting to and recording the decline of oyster populations and describing the collapse of fishing, but also of oyster reefs. Since very few of these accounts and papers provide information of ecosystem distribution within a location, we measured ecosystem decline as presence/absence of the ecosystem at each recorded historical location.

2.1.2. Criterion B: Restricted geographic distribution

For Criterion B we used ecosystem mapping and distribution data provided by Gillies et al. (2018), which compiles data from several studies (Diggles, 2013; Alleway and Connell, 2015; Gillies et al., 2015a; Warnock and Cook, 2015; Ford and Hamer, 2016; Jones and Gardner, 2016). These studies use methods such as side scan and multibeam sonar, GPS mapping, aerial photos, harvest reports and eyewitness accounts to determine current ecosystem distribution.

Area of Occupancy (AOO) was calculated from these data using a single point for each of the known locations in which the ecosystem is found. A 10 km grid map was constructed over the entire distribution of the ecosystem. We chose to include all grid cells even when the ecosystem occupied <1% because of the small patch sizes associated with the ecosystem (i.e. typically 100 m²–2000 m²). The level of uncertainty around this calculation is relatively high. The AOO map and calculations were performed using the GDA94/Geoscience Australia Lambert Projection. To calculate Extent of Occurrence (EOO), a minimum convex polygon (no internal angles are >180°) enclosing all the data was then created in ArcGIS.

2.1.3. Criterion C: Environmental degradation

For Criterion C, we used three variables to quantify environmental degradation of the ecosystem. Firstly, we use catchment land use as an indicator of the variable sediment load in estuaries (Chan et al., 2002; Taylor et al., 2004; Thrush et al., 2004). Increased sediment is known to be a primary inhibitor of oyster reef development and persistence, whereby high sediment loads can cause death by smothering, inhibiting oyster settlement or enhancing oyster parasites and disease such as mudworm (Ogburn et al., 2007; Fitzsimons et al., 2019). The eastern and southern coasts of Australia have undergone significant changes in land use since European settlement (Mansergh et al., 2006) and the causal impact this has had on altering river and estuary ecosystems is well known (Prosser et al., 2001).

To determine the level of severity in the indicator sediment load, we analysed the most current (2017) Catchment Scale Land Use of Australia data set (https://data.gov.au/dataset/catchment-scale-land-use-of-australia-update-2017). We used percentage of land use change within each associated catchment and applied the relevant IUCN thresholds: i.e. where more than 80% of the catchment had been classified as either ‘land for production use’ or ‘intensive use’ this corresponded to a high severity level for that location (i.e. high degree of environmental degradation) and a corresponding severity risk rating of Critically Endangered. Where more than 50% was classified as land under production or intensive use, we classified the location as having a severity risk rating of Endangered and where there was more than 30% of land for production use or intensive use this corresponded to a severity risk rating of Vulnerable. To determine threat extent, we use the proportion of catchments across the ecosystem’s distribution which contained a threat rating.

Secondly, we use extent of estuary shoreline modification as an indicator of substrate simplification. Modified shorelines can alter or remove abiotic conditions suitable for ecosystem growth and persistence (i.e. elevation, slope, wave energy dynamics, substrate type, availability of hard surfaces). We quantified the percent of shoreline loss by selecting a 2 km buffer around estuaries with historical reefs as the analysis area, then calculated the percentage of different land use types for each estuary. We calculated the percentage of land classified as nature conservation areas/minimal use and the percentage of land calculated as urban intensive uses (including residential, commercial buildings, transport infrastructure) for each estuary within this area. We assigned threat categories to each location using the same method described above.

Thirdly, to assess future threats, we identified the main drivers likely to affect biotic and abiotic interactions of oyster reefs from a broader list of key threatening processes for coastal and estuary systems identified from the Department of Climate Change (2009), Gillanders et al. (2011), Hobday and Lough (2011) and Clark and Johnston (2017) and derived the associated impact of the stressors on oyster reef ecosystems from the literature (see Historical and Current Threatening Processes section above, also summarised in Fig. 2 and Table 3).
Collectively, these three indicators were used to demonstrate plausible relationships between catchment change as the primary driver of several abiotic stressors that are known to affect ecosystem growth and persistence. These connections are highlighted in our ecosystem conceptual model (Fig. 2).

2.1.4. Criterion D: Disruption of biotic processes or interactions

For Criterion D, we selected the biotic indicator abundance of key species (oysters) as the primary mechanisms to assess decline in altered biotic interactions. Oyster reefs in their reference condition are characterised by high densities of oysters which provide habitat, shade, food and shelter for a diverse flora and fauna assemblage. Oysters are an ecosystem engineer and loss of oyster biomass to levels defined as a collapsed state (Table 1) disrupts fundamental biotic processes that sustain reef persistence and creation on which most reef-associated flora and fauna rely. Assessment was made by considering (qualitatively) the strength of the drivers identified in Criterion C against published data on the effect of biotic processes and interactions. Where the ratio of oyster recruitment through to reproductive or mature age is higher than oyster mortality, the ecosystem can feasibly exist in a steady state or expand in size and maintain a shell budget. Where recruitment is limited, or when oysters are unable to survive to maturity the ecosystem will either maintain a steady state, or where mortality exceeds recruitment, the ecosystem will decline.

2.1.5. Criterion E: Quantitative analysis that estimates the probability of ecosystem collapse

We did not conduct an assessment against Criterion E, due to the small and isolated number of remaining reefs each occurring in different estuary systems. The complex hydrodynamics associated with estuaries and coarse nature of the available time series data inhibits an ecosystem-wide quantitative assessment of future collapse over the time series of 50–100 years. We therefore classified Criterion E as Data Deficient.

3. Results

The assessment revealed different levels of threat detectable by different indicators from Data Deficient to Critically Endangered, with all of the four main criteria assessed having at least one of the three sub-criteria with a risk rating equalling Vulnerable (Table 2). Overall, the degree of confidence in the data varied (e.g. high degree for Criteria A, ‘since 1750’) to less confidence (e.g. Criteria D, ‘past 50 years’). Two of the four criteria assessed against the ‘Since 1750’ time category were assessed as Critically Endangered and these were consistent when analysed for both sub-community types. Both sub-community types had similar risk ratings, and met similar criteria for assessment as Critically Endangered, although the O. angasi sub-community was assessed as particularly high for several criteria (A,B) because of its highly restricted geographic distribution. Overall, and as per the IUCN Red List for Ecosystems methodology, taking the highest risk rating, SEA Oyster Reefs were assessed as Critically Endangered.

3.1. Criterion A: Reduction in geographic distribution

3.1.1. A1: Reduction in the past 50 years

Gillies et al. (2018) identified seven locations which currently contain SEA Oyster Reefs, only one of which contains the O. angasi community sub-type. Two additional locations for O. angasi community sub-type have recently been identified in
Tasmania and Victoria (pers. obs) but these have yet to be assessed. Likewise, in New South Wales, anecdotal evidence exists for *S. glomerata* reefs in other locations (NSW DPI, 2019) not identified by Gillies et al. (2018), yet these have yet to be mapped or verified as reef ecosystems. Current verified best estimates therefore indicate that only seven (but potentially nine) of an estimated 178–303 (lower-upper estimates, Gillies et al., 2018) historical locations (i.e. bays, estuaries, embayments) contain a remnant of the ecosystem (inclusive of both community sub-types).

Ford and Hamer (2016) provide evidence that limited oyster harvesting (30 tonnes per year) still occurred in Port Phillip, Victoria, up until the mid-twentieth century indicating that reefs or dense beds were still present around 50 years ago in that region, although the extent to which these were *O. angasi* compared to *Mytilus (edulis) galloprovincialis* (blue mussel) which were also harvested at the time is unknown. In all other locations, reports of collapse for the ecosystem had occurred prior to 1950 (Kirby, 2004; Diggles, 2013; Alleway and Connell, 2015; Ford and Hamer, 2016; Gillies et al., 2018; Thurstan et al., 2020). Despite the potential for decline within the last 50 years, evidence of recent loss is limited and further work needs to be undertaken to address this knowledge gap. We therefore conclude that the status of the ecosystem under this sub-criterion (i.e. past 50 years) is Data Deficient.

### 3.1.2. A2: Reduction over the next 50 years

We infer the risk of future ecosystem collapse over the next 50 years will be based largely on the extent to which further environmental degradation occurs, since the primary historical stressors (harvesting through dredge methods, massive land use change) have largely abated and are unlikely to reoccur in all Australian states where the ecosystem is found. There was also insufficient data to project a quantitative estimate of the future distribution and we assess the status here as Data Deficient.

#### 3.1.3. A3: since 1750

Gillies et al. (2018) described the decline in the *O. angasi* sub-community from 118 historical locations (most conservative estimate) to just one location known today, a decline of over 99%. For *S. glomerata* community sub-type, only 6 of 60 historical locations (conservative estimate) have been identified, resulting in a 90% decline. Collectively for the ecosystem, the most optimistic national assessment indicates seven of 178 historically known locations still occur today, resulting in a decline of 94%. Gillies et al. (2018) conclude that ecosystem decline occurred primarily over a 150-year period from 1800 to 1950 which coincided with the peak wild oyster harvest fishery, landscape modification for the primary purpose of agriculture, forestry and urbanization, and industrialization of coastal areas and estuaries across south-eastern Australia (Gillies et al., 2015a, 2018).

Kirby (2004) described the collapse of all natural oyster fisheries (primarily *S. glomerata*) in New South Wales and southeast Queensland by 1910 which is similar to Ogburn et al.’s (2007) estimate that New South Wales subtidal oyster reefs (primarily *S. glomerata*) were in decline by 1880. In Victoria, Ford and Hamer (2016) describe >90% loss of *O. angasi* reefs in Port Phillip, Western Port and Corinner Inlet coastal systems by 1860, although oyster fisheries were able to continue at much lower biomass until 1970. Alleway and Connell (2015) describe a collapse of the *O. angasi* fishery and reefs across at least 1500 km of coastline in South Australia by 1944. Warnock and Cook (2015), describe the loss of oyster beds (*O. angasi*) in southwest Western Australia estuaries by 1940. At the estuary scale, Diggles (2013) describes collapse of subtidal *S. glomerata* communities by 1920 and Edgar and Samson (2004) indicate a 100% decline of *O. angasi* beds in the D’entrecasteaux Channel, Tasmania, by 1930.

Based on the weight of evidence from the above studies, the rate of ecosystem decline after European settlement was rapid and directly associated with an increase in commercial harvest which had largely ceased across the ecosystem’s distribution by 1920. We therefore assess the status of the ecosystem under sub-criterion A3 as Critically Endangered (including for both community sub-types) with a high degree of confidence.

### 3.2. Criterion B: Restricted geographic distribution

#### 3.2.1. B1: Extent of Occurrence

The minimum convex polygon encompassing all confirmed remaining sites (7) encompasses an Extent of Occurrence (EOO) of 73,250 km² (Fig. 3), which, when using the process of Bland et al. (2017), is considered as ‘Least Concern’. We also re-ran the assessment separately for the *S. glomerata* sub-community which provided an EEO of 47,541 km². The current single location known for *O. angasi* would equal an EEO of <1 km² (Keane and Gardner, 2018).

Sub-criteria B1–B3 also requires an assessment of the number of threat-defined locations (defined as a geographically or ecologically distinct area in which a single threatening event can rapidly affect all occurrences of an ecosystem type; Bland et al., 2017). For the *O. angasi* sub-community, only a single population is known to occur in north-eastern Tasmania making this sub-community type extremely vulnerable to single catastrophic events such as floods, droughts, storms, and potentially, recruitment failure if the existing commercial oyster fishery were to cause local depletions (Keane and Gardner, 2018). We therefore categorized this region as a single threat-defined location. For the *S. glomerata* sub-community, populations in New South Wales and south-eastern Queensland can be exposed to single catastrophic events across the entire region (specifically land and marine heatwaves and droughts) but also other events which can affect one or more catchments (e.g. east coast flooding, hypoxic black water events) at one time but are unlikely to affect the entire ecosystem extent. From a management
view, in New South Wales, all reef locations are located within the Coastal Vulnerability Area, a spatial zone defined under the New South Wales Coastal Management Act 2016, which is identified largely because it has the same coastal threats and vulnerability. Regardless of whether one threat-defined location (entire region-heatwaves and droughts) or six threat locations (catchments-floods and blackwater events) are identified, the risk rating would be the same (i.e. ≤ 10, Vulnerable). When considering the EOO for the SEA Oyster Reefs (comprising both community sub-types) the ecosystem was classified as Least Concern (Extent of Occurrence is > 50,000 km²). When considering the community sub-types individually the S. glomerata community sub-type was classified as Vulnerable and, based on the extremely low EOO and single threat-location, we classified the O. angasi community sub-type as Critically Endangered.

3.2.2. B2: Area Of Occupancy (AOO)

We identified seven out of a total of 193 cells (3.6%) as occupied by the ecosystem (Fig. 4), although in several estuaries the area of occupancy is likely to only occupy <1% of the grid cell (i.e. less than 1 km² as indicated by McLeod et al. (2020), demonstrating the ecosystem is currently severely fragmented. Yet because of the uncertainty of the total area occupied at each location (i.e. not all reef patches were mapped at each location by McLeod et al. (2020) we were cautious and included all cells within our assessment. A grid count of seven cells indicates a risk rating of Endangered (≤20 cells and less than five threat locations, included – see B1 above). We therefore assess the risk rating for B2 as Endangered, with S. glomerata community sub-type (6 grid cells) assessed as Endangered and O. angasi (1 cell) Critically Endangered.

3.2.3. B3: Number of threat-defined locations

The ecosystem can be considered Vulnerable (the only threat category available in this sub criterion) because it meets the criteria of occurring in less than five threat-defined locations and both community sub-types are vulnerable to complete collapse from single catastrophic events (described above) which could occur in the immediate future and over a short period of time.

3.3. Criterion C: Environmental degradation

3.3.1. C1: The past 50 years

Due to the difficulty in linking drivers and threats relating to biotic degradation across the ecosystem’s entire extent to the past 50 year time horizon only, we were unable to complete an analysis for this sub-criterion and we classified this as Data Deficient.
Fig. 4. Current extent and historical distribution and Area of Occupancy of the Oyster Reef Ecosystem of Southern and Eastern Australia, with each cell representing 10 sq km. Coastal embayments in blue represent potential ecosystem occupation. Data derived from Gillies et al. (2018). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)
3.3.2. **C2: The next 50 years**

Of the 25 coastal and estuary drivers and threats identified in Australia by Clark and Johnston (2017), seven have the potential to cause ecological collapse of SEA Oyster Reefs by altering conditions that control both the abiotic and biotic conditions required for ecosystem persistence (Table 3). All but one of these threats (low-oxygen dead zones) are expected to deteriorate further in the future in southern and eastern Australia (Gillanders et al., 2011; Clark and Johnston, 2017), posing a higher risk of collapse to the ecosystem compared with today. In particular, ‘climate and weather’ is considered to have a ‘very high impact’ on Australia’s bays and estuaries, with mean annual rainfall expected to decrease, storm events increase and sea level rise expected to be higher for south-eastern Australia compared to the global average (Department of Climate Change, 2009; Hobday and Lough, 2011; McInnes et al., 2016). These current and future threats which are expected to increase in intensity in the near future, provide a high level of confidence that the ecosystem is at risk of future collapse within the next 50 years. However, we assessed this criterion as Data Deficient because whilst there is certainty that threats will increase in the near future and are likely to have an impact on oyster populations, we were unable to determine the likely adaptive response of the ecosystem (see Discussion).

3.3.3. **C3: since 1750**

The relative severity of catchment modification as a driver of the abiotic stressor sediment supply equated to threat rankings ranging from Least Concern to Critically Endangered, with 90% of all extant sites (n = 198) assessed as Vulnerable or higher (Table 4). This resulted in a Vulnerable risk rating for this indicator because the assessment meets the threshold of >30% degradation across >80% of the ecosystem extent. Similarly, 86% of catchments (n = 178) assessed for estuary shoreline modification as a driver of substrate simplification had a high degree (>50% degradation, Table 4), resulting in an overall risk rating of Vulnerable. Collectively for these two indicators our assessment suggests a plausible threat of historical environmental degradation as a result of catchment and estuary shoreline modification across most of the extent of the ecosystem. We thus assessed this criterion as Vulnerable with a high degree of certainty.

### 3.4. Criterion D: Disruption of biotic processes

3.4.1. **D1: The past 50 years**

Only one study in a single location (Sydney) has observed an increase in natural oyster (S. glomerata) abundance over the last 50 years (Birch et al., 2014). In aquaculture, oyster production has declined by half since peak production in mid 1970s, in part attributed to disease (QX, Winter Mortality Syndrome) and declining water quality (White, 2001; NSW DPI, 2016). Whilst these drivers are likely to also have affected wild oyster populations, unfortunately, no similar long-term assessment of wild oyster populations or recruitment have been published. We therefore assess this sub criteria as Data Deficient.

3.4.2. **D2: The next 50 years or any 50-year period**

Projections of distribution and biomass of oyster ecosystems in the next 50 years are limited and the status of the ecosystem under this sub-criterion was considered Data Deficient.

3.4.3. **D3: Since 1750**

Several studies have previous documented collapse of the ecosystem as a result of oyster extraction (Ogburn et al., 2007; Lergessner, 2008; Diggles, 2013; Alleway and Connell, 2015; Gillies et al., 2015a, 2018; Ford and Hamer, 2016, Thurstan et al., 2020), though only three have quantified decline in oyster abundance or biomass. Thurstan et al. (2020) document total

<table>
<thead>
<tr>
<th>Location</th>
<th>% catchment modification</th>
<th>Catchment rating</th>
<th>% estuary shoreline modification</th>
<th>Shoreline rating</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moreton Bay, Queensland</td>
<td>7</td>
<td>LC</td>
<td>75</td>
<td>EN</td>
</tr>
<tr>
<td>Richmond River, New South Wales</td>
<td>67</td>
<td>EN</td>
<td>76</td>
<td>EN</td>
</tr>
<tr>
<td>Port Stephens, New South Wales</td>
<td>49</td>
<td>VU</td>
<td>25</td>
<td>LC</td>
</tr>
<tr>
<td>Hunter River, New South Wales</td>
<td>64</td>
<td>EN</td>
<td>79</td>
<td>EN</td>
</tr>
<tr>
<td>Botany Bay, New South Wales</td>
<td>77</td>
<td>EN</td>
<td>89</td>
<td>CR</td>
</tr>
<tr>
<td>Crookhaven River, New South Wales</td>
<td>46</td>
<td>VU</td>
<td>34</td>
<td>VU</td>
</tr>
<tr>
<td>Georges Bay, Tasmania</td>
<td>42</td>
<td>VU</td>
<td>34</td>
<td>VU</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>All current and previously known sites (% of total)</th>
<th>% catchment modification</th>
<th>Catchment rating</th>
<th>% estuary shoreline modification</th>
<th>Shoreline rating</th>
</tr>
</thead>
<tbody>
<tr>
<td>(N = 198)</td>
<td>15.2 (30)</td>
<td>CR</td>
<td>19.1 (34)</td>
<td>CR</td>
</tr>
<tr>
<td>(N = 178)</td>
<td>39.4 (78)</td>
<td>EN</td>
<td>51.1 (91)</td>
<td>EN</td>
</tr>
<tr>
<td>35.4 (70)</td>
<td>VU</td>
<td>15.7 (28)</td>
<td>VU</td>
<td></td>
</tr>
<tr>
<td>10.1 (20)</td>
<td>LC</td>
<td>14.0 (25)</td>
<td>LC</td>
<td></td>
</tr>
</tbody>
</table>
collapse of the ecosystem, estimating a 96% decline in *S. glomerata* fisheries production in 2016 compared to the peak of the fishery in 1891. Allaway and Connell (2015) document a similar (96%) decline in harvest records for *O. angasi* between 1886 and 1944 and Ogburn et al. (2007) indicate a 66% decline in both *S. glomerata* and *O. angasi* production in New South Wales.

It was relatively common for historical accounts to describe vast oyster systems ranging from several hundred square meters in length to several kilometers which were intensively harvested for oysters. For instance:

The Fisheries Inspector for Moreton Bay, Fison (1884) reported for *S. glomerata* in Pumicestone Passage, Queensland:

“33 thousand bags of oysters have been taken, they being in some places four and five feet deep, Mr Freeman having informed me that he has made his boat fast to a stake, and dredged for six weeks”

The New South Wales, Royal Commission on Oyster Culture (1876-7) (New South Wales, 1877) reported for *S. glomerata* in Port Stephens, New South Wales:

“In the 1860’s a man could work his warp stake into the bed and not leave that spot for sixteen or twenty days, getting fifteen to twenty bags a day all that time. For a long time ten to twelve or even fifteen boats were so employed until only three or four bags could be got… some came back in about three years only to get at most six or seven bags per day”.

The Illustrated Australian News (Anon.) (7 November 1891, pp. 8–9) reported for *O. angasi* in Port Albert, Victoria:

“An account of oyster dredging offshore from Corner Inlet describes an oyster bank ‘from Shallow Inlet towards Wilson Promontory for a distance of 12 miles’ and another ‘3 miles long beginning at the (Corner) Inlet’”

Harvest records, whilst not comprehensive, provide an insight into the extent of oyster biomass (typically mature oysters) extracted during previous commercial harvest. For instance, in Western Port, Victoria, during the mid-1850s, 1.2 million dozen oysters were removed per year (Ford and Hamer, 2016). In southeast Queensland, harvest records began in 1874, with an estimated peak in 1891 recording removal of 2–3.65 million dozen oysters per year (Thurstan et al., 2020) and in South Australia, during the 1880s over 100,000 dozen oysters were harvested per year (Alleway and Connell, 2015). Further examples from individual estuaries and industries can be found in Gillies et al. (2015a, 2018, Table 3) and Thurstan et al. (2020). In all circumstances the wild harvest industry collapsed, which often prompted early attempts at aquaculture and ranching (e.g. the laying down of oysters or substrate; Roughly, 1922), before modern cage aquaculture begun in the early 1950s.

A lack of modern data on oyster densities from historical locations and quantitative estimates on historical abundance or biomass precluded a quantitative assessment of decline for known sites. Nonetheless, since the above studies describe extensive reef systems that were intensively harvested across the entire extent of the ecosystem and with most of these studies concluding total collapse of the ecosystem primarily as a result of oyster harvest, we believe there is sufficient evidence to reasonably deduce that the relative severity related to loss of oyster biomass causing ecosystem decline is $\geq 90\%$ of past biomass (all studies indicate ecosystem collapse through loss of oysters) and the extent of threat was $\geq 90\%$ of past extent (studies cover the full geographic range of the ecosystem). We therefore assessed the status of the ecosystem under this sub-criterion as Critically Endangered with a medium degree of confidence.

4. Discussion

4.1. Status of the Oyster Reef Ecosystem of Southern and Eastern Australia

We assessed the conservation status of the Oyster Reef Ecosystem of Southern and Eastern Australia using the IUCN framework and determined that the ecosystem (including both community sub-types) should be classified as Critically Endangered, the most severe risk rating. Risk ratings ranged across all threat category types, from a Critically Endangered assessment given for Criterion A: Reduction in Extent (since 1750) and Criterion D: Disruption of Biotic Processes (since 1750), through to Least Concern for Criterion D: Disruption of Biotic Processes (past 50 years). Overall the ecosystem met the listing requirements for all criteria (with the exception of Criterion E: Quantitative Analysis, which we were unable to assess), but not for all sub-criteria. The level of confidence also varied among and within criteria. For instance, we found sufficient evidence to quantify the decline in ecosystem extent and oyster biomass throughout the 1800s (largely due to the well documented decline in the wild oyster harvest industry), yet there was little information on the extent of decline over the last 50 years. This result validated Alleway and Connell (2015) observations of shifting baselines for shellfish ecosystems related to loss of memory in recent generations where the general visibility and awareness of oyster reef ecosystems, predominantly over the past 50 years, has been low.

Our assessment and the IUCN Red List process may be of value for other shellfish ecosystems, particularly those that are likely to have undergone significant decline (Beck et al., 2011) and are actively being restored, such as *O. edulis* in Europe, *O. chilensis* and *Perna canaliculus* in New Zealand, *C. virginica* and *O. lurida* in the United States and *C. hongkongensis* in Hong Kong (Fitzsimons et al., 2019). A detailed understanding of ecosystem definition, collapse thresholds and ecological risks can help to inform priority locations for protection and restoration and assist with developing methods for restoration by describing key ecosystem functions and structural attributes, which can guide the development of reference models (Gillies et al., 2017). Even if an ecosystem assessment does not meet any risk thresholds, undertaking the process itself can reveal new insights into the ecosystem (including gaps in understanding), and if undertaken regularly, can be used to monitor the status of the ecosystem over time (Alaniz et al., 2019). We also surmise the high ecological value yet relatively small and patchy...
nature of the ecosystem is representative of the concept of ‘small natural features’ such as desert springs, bat caves, temporary pools and coral heads which represent management challenges but also novel opportunities for their protection and restoration (Hunter et al., 2017).

A significant gap in our understanding of this ecosystem is how it will respond to future threats, particularly from climate change. Stressors such as altered water flow, salinity, hypoxia, heat stress and ocean acidification are already increasing or expected to increase in Australian estuaries (McInnes et al., 2016; Clark and Johnston, 2017), yet there is an insufficient number of studies that can confidently predict how estuarine, and particularly shellfish, ecosystems are likely to respond (but see Watson et al., 2009; Gillanders et al., 2011; McAfee et al., 2017; Parker et al., 2017). SEA Oyster Reefs have the potential to migrate within an estuary and could conceivably colonise new estuaries to avoid stress and remain within physiological thresholds, but this is assuming sufficient substrate and oyster biomass is available for local recruitment, settlement and reef creation. We suggest future research should prioritise the development of climate response ecosystem models to understand whether changing climatic factors will exacerbate the identified risk of total collapse and to help identify areas for future protection and management.

There is growing anecdotal evidence that a number of unmapped S. glomerata reefs may exist in New South Wales (NSW DPI, 2019) which still require verification as oyster reefs. Oyster reefs may also be establishing on abandoned oyster leases and these are the focus of new restoration sites by the NSW Government (Kylie Russell, NSW DPI, pers. comm.). We suggest that verification of S. glomerata reefs in NSW should be prioritised and the S. glomerata community sub-type subsequently re-assessed. We note though that in order for the assessment to downgrade from its current assessment as Critically Endangered for Criteria A, the number of validates sites would need to more than double (>15), but this would likely not affect the over rating of Critically Endangered due to the significant historical loss.

4.2. Implications for conservation listing

The assessment of SEA Oyster Reefs as Critically Endangered, has implications for listing under environmental legislation in Australian jurisdictions. Listing under threatened species/communities or related legislation confers a number of important benefits to the ecosystem. These benefits can include: 1) preventing or limiting direct physical destruction/degradation of the system, 2) recognising, listing or addressing threatening processes that might be having an indirect role in degradation, and 3) prioritising and financing conservation and restoration activities related to the ecosystem. Furthermore, the assessment process can assist in identifying appropriate conservation policies that address specific ecosystem risks highlighted by each criterion (Alaniz et al., 2019).

Australia is a federated nation, and environmental law rests primarily with the states and territories (sub-national governments), with some overlapping national government responsibilities (such as for nationally threatened ecological communities). As such, and as there were no distinct differences in our threat assessments between states, listing SEA Oyster Reefs under relevant legislation should be a high priority. Not all Australian states have legislation that enables listing of threatened marine ecological communities. The most relevant current legislation for listing is as follows: Australian Government (national level) — Environment Protection and Biodiversity Conservation Act 1999 (listing application accepted for assessment in 2018); Western Australia — Biodiversity Conservation Act 2016; South Australia — Fisheries Management Act 2007; Victoria — Flora and Fauna Guarantee Act 1988; Tasmania — Nature Conservation Act 2002; New South Wales — Biodiversity Conservation Act 2016; and Queensland — Nature Conservation Act 1992 (critical habitat listing).

Despite the risk of ecosystem collapse for SEA Oyster Reefs, shellfish reefs may be one of the most restorable marine ecosystems globally. Australia’s coastal environments have experienced extensive environmental change over the past 200 years, yet Australia’s east coast oyster populations have demonstrated resilience to environmental stressors (e.g. McAfee et al., 2017) and readily adhere to most hard substrates. Restoration efforts in Australia and the United States demonstrate that through active restoration methods including the addition of settlement substrate and oyster larvae, many 100s of hectares can be restored within single systems (Schulte et al., 2009; Fitzsimons et al., 2019; https://www.shellfishrestoration.org.au/). The environmental, economic and social benefits of undertaking such restoration are well documented (Coen et al., 2007; Grabowski et al., 2012; Kroeger, 2012; McLeod et al., 2019) and interest in scaling-up marine ecosystem restoration is growing (e.g. Fitzsimons et al., 2015; Gillies et al., 2015b). These studies and the prominent risk of total collapse identified in this study provide a compelling case for new investment that can arrest, and potentially reverse, the decline of the ecosystem.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References


Urgent Need to Use and Reform Critical Habitat Listing in Australian Legislation in Response to the Extensive 2019–2020 Bushfires

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The unprecedented bushfires of 2019–2020 burnt significant areas of forest and woodland in eastern and southern Australia. The size of the area burnt, and intensity of the fire has meant the majority of habitat for a large number of threatened species has been impacted, placing high conservation value on the unburnt refuges remaining. Most Australian jurisdictions have provision for “critical habitat” listing under their threatened species legislation. However, these provisions have been under-utilised. Here, I review these provisions in the jurisdictions impacted by these fires. Considering the number of threatened species and the crucial role of critical habitat in their recovery, new processes will need to be implemented to rapidly assess and designate critical habitat under existing provisions and future reforms to legislation implemented in order to deal with future events such as these extensive bushfires.

INTRODUCTION

The bushfires of the spring-summer of 2019–2020 burnt an estimated 10 million ha in New South Wales, the Australian Capital Territory, Victoria, Tasmania, south-east Queensland, southern South Australia and south-west Western Australia. Not only is the scale of Australian forest and temperate woodland burnt in a single season unprecedented, the combination of the intensity of the fires and the very large individual fire scars has resulted in a significant loss of biodiversity and, in many places, forest and woodland structure. Estimates of the number of land mammals, birds and reptiles killed is conservatively estimated at one billion in New South Wales and Victoria,1 and many times greater for all species.2 Critically, the fires have had significant impacts on the habitat for a range of threatened species. Some 49 listed threatened species have more than 80% of their modelled likely or known distribution within the fire extent and a further 65 listed threatened species have more than 50%, but less than 80%, of their modelled likely or known distribution within the fire extent.3

The Commonwealth Government’s Wildlife and Threatened Species Bushfire Recovery Expert Panel identified “Protecting unburnt areas within or adjacent to recently burnt ground that provide refugia” as one of the top priority activities to prevent extinctions and maximise the changes of long-term recovery

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of native species and communities and the importance of refugia has also been recognised by State government responses (e.g. Victoria and New South Wales).

Most government responses to the bushfires to date have focused on important management actions. Considering the importance of this refugia for the future survival for a range of threatened taxa and similar provisions within conservation legislation at State and Federal level. I briefly outline how critical habitat is considered in legislation in jurisdictions impacted by the 2019–2020 fires of eastern and southern Australia, and then discuss the opportunity to better utilise existing provisions following the immediate bushfire crisis and the need to make these provisions more relevant to emerging conservation challenges.

CRITICAL HABITAT PROVISIONS IN LEGISLATION OF FIRE-AFFECTED JURISDICTIONS

Federal

Under the Environment Protection and Biodiversity Conservation Act 1999 (Cth), the Minister may list habitat identified by the Minister as being critical to the survival of a listed threatened species or listed threatened ecological community on a Register of Critical Habitat. Considerations for listing are further defined in the Environment Protection and Biodiversity Conservation Regulations 2000 (Cth), and includes use “in periods of stress”, including fire. In considering whether to list habitat, the Minister must take into account the potential conservation benefit of listing the habitat and the regulations must require the Minister to consider scientific advice (from the Threatened Species Scientific Committee) in identifying the habitat. The Minister must, when making or adopting a recovery plan, consider whether to list habitat that is identified in the recovery plan as being critical to the survival of the species or ecological community for which the recovery plan is made or adopted. Before listing habitat in the register, the Minister must be satisfied that reasonable steps have been taken to consult with the owner of the property where the habitat is located, if the habitat is not in a Commonwealth area. While it is an

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7 Environment Protection and Biodiversity Conservation Act 1999 (Cth) s 207A (1).

8 Environment Protection and Biodiversity Conservation Regulations 2000 (Cth) s 7.09 (1). Specifically, the Minister may take into account the following matters: (1) whether the habitat is used during periods of stress (Examples of period of stress: Flood, drought or fire); (2) whether the habitat is used to meet essential life cycle requirements (Examples: Foraging, breeding, nesting, roosting, social behaviour patterns or seed dispersal processes); (3) the extent to which the habitat is used by important populations; (4) whether the habitat is necessary to maintain genetic diversity and long-term evolutionary development; (5) whether the habitat is necessary for use as corridors to allow the species to move freely between sites used to meet essential life cycle requirements; (6) whether the habitat is necessary to ensure the long-term future of the species or ecological community through reintroduction or re-colonisation; (7) any other way in which habitat may be critical to the survival of a listed threatened species or a listed threatened ecological community.

9 Environment Protection and Biodiversity Conservation Act 1999 (Cth) s 207A (1A).

10 Environment Protection and Biodiversity Conservation Act 1999 (Cth) s 207A (2).

11 Environment Protection and Biodiversity Conservation Regulations 2000 (Cth) s 7.09(2).

12 Environment Protection and Biodiversity Conservation Regulations 2000 (Cth) s 7.09(3)(b).
offence to knowingly damage critical habitat, this only applies to habitat that is in or on a Commonwealth area. Only five areas are listed on the Register of Critical Habitat, and none since 2005.

Victoria

Victoria’s primary legislation for threatened species, the Flora and Fauna Guarantee Act 1988 (Vic), has recently gone through a review process and will be amended when the Flora and Fauna Guarantee Amendment Act 2019 (Vic) comes into effect on 1 June 2020. The Victorian Government has acknowledged it has not used the critical habitat mechanism in the current Act “…for a range of reasons, including concerns over the regulatory burden placed on landholders and the scientific challenges of identifying critical habitats and determining their boundaries.” Critical habitat has only been declared once (in 1996) and was subsequently revoked a year later, while Interim Conservation Orders have never been used.

Under the amended Act “The Secretary may determine any area of Victoria to be a critical habitat if the area significantly contributes to a) the conservation in Victoria of a listed taxon or community of flora or fauna, b) a taxon or community of flora or fauna that is not listed but recommended by the Scientific Advisory Committee or c) the area supports ecological processes or ecological integrity that significantly contributes to the conservation of a taxon or community that is listed”. Further factors influencing determining of critical habitat are outlined, including “refugia during environmental stress”.

Before making a critical habitat determination, the Secretary must give written notice to the following: (1) the landholder of any land that is within the area of the proposed determination; (2) any public authority that performs a function or exercises a power in the area of the proposed determination; (3) any person whose interests, in the opinion of the Secretary, are likely to be adversely affected by the proposed determination, although this does not apply if the Secretary considers that the written notice of a proposed determination is likely to result in damage being done to the habitat within the area that is subject to the determination.

Interim Conservation Orders will become Habitat Conservation Orders under the amended Act. Habitat Conservation Orders prohibit or regulate activities or processes that take place within Critical Habitat, or, if it adversely affects that habitat, any activity that takes place outside that habitat. A habitat conservation order remains in force for the period specified in the order which must not exceed 10 years after the order takes effect. It is an offence to contravene a habitat protection order, and compensation is payable to landowners for financial loss suffered as a consequence of the making of a habitat conservation, with the amount determined by the Secretary.

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13 Environment Protection and Biodiversity Conservation Act 1999 (Cth) s 207B(1)(c).
16 DELWP, n 15.
17 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 15, referring to amendments to s 20 of the Principal Act.
18 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 15, referring to amendments to s 20 of the Principal Act.
19 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 15, referring to amendments to s 20B of the Principal Act.
20 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 15, referring to amendments to s 20B of the Principal Act.
21 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 20, referring to amendments to s 27(1) of the Principal Act.
22 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 20, referring to amendments to s 27(1) of the Principal Act.
23 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 20, referring to amendments to s 32 of the Principal Act.
24 Flora and Fauna Guarantee Amendment Act 2019 (Vic) s 20, referring to amendments to s 39 of the Principal Act.
New South Wales

Under the New South Wales *Biodiversity Conservation Act 2016* (NSW), the Minister may declare any area in New South Wales to be an area of outstanding biodiversity value\(^{25}\) by publication of a notice of the declaration on the NSW legislation website.\(^{26}\)

Areas of declared critical habitat under the *Threatened Species Conservation Act 1995* (NSW) have become the first Areas of Outstanding Biodiversity Value in New South Wales with the commencement of the *Biodiversity Conservation Act 2016* (NSW).\(^{27}\) Only four Area of Outstanding Biodiversity Value declarations have been made (for Gould’s Petrel, Little Penguin population in Sydney’s North Harbour, Mitchell’s Rainforest Snail in Stotts Island Nature Reserve, and for Wollemi Pine).\(^{28}\)

An area may be declared as an area of outstanding biodiversity value if the Minister is of the opinion that (1) the area is important at a State, national or global scale, and (2) the area makes a significant contribution to the persistence of at least one of the following: (a) multiple species or at least one threatened species or ecological community, (b) irreplaceable biological distinctiveness, (c) ecological processes or ecological integrity, (d) outstanding ecological value for education or scientific research.\(^{29}\)

The declaration of an area may relate to, but is not limited to, protecting threatened species or ecological communities, connectivity, climate refuges and migratory species.\(^{30}\)

Before an area is declared to be an area of outstanding biodiversity value, the Environment Agency Head must (1) recommend the declaration of the area, (2) notify landholders whose land is within the proposed area, and any public authorities that appear to the Agency Head to exercise functions in relation to land within the proposed area, of the recommendation to declare the area, (3) give those landholders and public authorities a reasonable opportunity to make submissions with respect to the recommendation to declare the area and (4) must seek and consider the advice of the Threatened Species Scientific Committee, the Biodiversity Conservation Trust and the Biodiversity Conservation Advisory Panel.\(^{31}\)

The Environment Agency Head must notify any landholder or public agency whose land is within the proposed area after declaration and take reasonable steps to enter into a private land conservation agreement with the landholder.\(^{32}\) A person who damages a declared area of outstanding biodiversity value is guilty of an offence.\(^{33}\)

Queensland

The Queensland *Nature Conservation Act 1992* (Qld) defines critical habitat as “habitat that is essential for the conservation of a viable population of protected wildlife or community of native wildlife, whether or not special management considerations and protection are required” and “may include an area of land that is considered essential for the conservation of protected wildlife, even though the area is not presently occupied by the wildlife”. A management principle of protected wildlife under the Act is to identify the wildlife’s critical habitat and conserve it to the greatest possible extent.\(^{34}\) The chief executive must keep a register of critical habitats.\(^{35}\)

\(^{25}\) *Biodiversity Conservation Act 2016* (NSW) s 3.1(1).

\(^{26}\) *Biodiversity Conservation Act 2016* (NSW) s 3.1(2).


\(^{28}\) NSW Government, n 27.

\(^{29}\) *Biodiversity Conservation Act 2016* (NSW) s 3.2 (1).

\(^{30}\) *Biodiversity Conservation Act 2016* (NSW) s 3.2 (2).

\(^{31}\) *Biodiversity Conservation Act 2016* (NSW) s 3.3.

\(^{32}\) *Biodiversity Conservation Act 2016* (NSW) s 3.4.

\(^{33}\) *Biodiversity Conservation Act 2016* (NSW) s 2.3.

\(^{34}\) *Nature Conservation Act 1992* (Qld) s 73(a)(iv).

\(^{35}\) *Nature Conservation Act 1992* (Qld) s 133(1)(b).
The Minister may prepare a conservation plan for any native wildlife, class of wildlife, native wildlife habitat or area that is, in the Minister’s opinion, an area of major interest.\(^{36}\) A conservation plan may make provision for the use or development of land, and activities, in an area identified under the conservation plan as, or including, a critical habitat or an area of major interest.\(^{37}\)

If the Minister is of the opinion that a critical habitat is subject to a threatening process that is likely to have significant detrimental effect on the habitat, the Minister may make an interim conservation order for the conservation, protection or management of the habitat.\(^{38}\) An interim conservation order cannot be in place for more than 60 days, although the Governor in Council may, by gazette notice, extend the order by not more than 90 days.\(^{39}\) If a critical habitat is the subject of an interim conservation order, regulation, conservation plan, the landholder of which these mechanisms are subject to is entitled to be paid by the State the reasonable compensation because of the restriction or prohibition.\(^{40}\)

**Australian Capital Territory**

The Australian Capital Territory’s *Nature Conservation Act 2014* (ACT) defines critical habitat as a “habitat that is critical to the survival of a species or ecological community”. In developing an action plan for threatened species or ecological community listed under the Act, critical habitat (if known) for species or communities is identified.\(^{41}\) There is little further reference to critical habitat in the Act in anything other than an action plan, a document that may not be binding on decision makers.\(^{42}\)

**South Australia**

South Australia’s *National Parks and Wildlife Act 1972* (SA) does not have critical habitat provisions, but Sch 1 of the *Native Vegetation Act 1991* (SA) states that native vegetation should not be cleared if, in the opinion of the Native Vegetation Council (1) it comprises a high level of diversity of plant species; or (2) it has significance as a habitat for wildlife; or (3) it includes plants of a rare, vulnerable or endangered species; or (4) the vegetation comprises the whole, or a part, of a plant community that is rare, vulnerable or endangered; or (5) it is significant as a remnant of vegetation in an area which has been extensively cleared; or (6) it is growing in, or in association with, a wetland environment [among a range of other provisions].\(^{43}\)

**Western Australia**

Under the Western Australian *Biodiversity Conservation Act 2016* (WA),\(^{44}\) habitat is eligible for listing as critical habitat if (1) it is critical to the survival of a threatened species or a threatened ecological community; and (2) its listing is otherwise in accordance with the ministerial guidelines.\(^{45}\) The CEO

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\(^{36}\) *Nature Conservation Act 1992* (Qld) s 120H(1).

\(^{37}\) *Nature Conservation Act 1992* (Qld) s 120H(5).

\(^{38}\) *Nature Conservation Act 1992* (Qld) s 102.

\(^{39}\) *Nature Conservation Act 1992* (Qld) s 105.

\(^{40}\) *Nature Conservation Act 1992* (Qld) s 108; *Nature Conservation Act 1992* (Qld) s 137A.

\(^{41}\) *Nature Conservation Act 2014* (ACT) s 100.


\(^{43}\) *Native Vegetation Act 1991* (SA) Sch 1, 1 – Principles of clearance of native vegetation.

\(^{44}\) Which came into effect in its full form on 1 January 2019. The Act replaces the *Wildlife Conservation Act 1950* (WA) which did not have critical habitat provisions.

must take reasonable steps to give written notice of the proposed listing, amendment or repeal to the
owner or occupier of the land on which the habitat is located.\textsuperscript{46} A register of critical habitats must be
established and maintained.\textsuperscript{47}

The CEO may give a written “habitat conservation notice” to a person requiring the person to ensure that
habitat damage, or further habitat damage, does not occur on land, if the CEO reasonably believes that
– (1) habitat damage is likely to occur on the land; or (2) habitat damage is occurring or has occurred on
the land.\textsuperscript{48} A habitat conservation notice may require any person bound by it to take specified measures,
within or for the duration of a specified period, for a range of purposes including (1) to repair any habitat
damage that has occurred and (2) to re-establish and maintain critical habitat in any area affected by
habitat damage to a condition as near as possible to the condition of the critical habitat before the habitat
damage occurred.\textsuperscript{49} The CEO must, by written notice given to the person, invite the person to make
submissions on the determinations of the habitat conservation notice.\textsuperscript{50}

The habitat conservation notice binds each successive owner or occupier of the land.\textsuperscript{51} A person who is
bound by a habitat conservation notice must not contravene the notice and penalties vary depending on
threat status of the species or ecological communities involved.\textsuperscript{52} The costs of taking measures on
the land in compliance with a habitat conservation notice are to be borne in the proportions determined by
the CEO.\textsuperscript{53}

\textbf{Tasmania}

The Tasmanian \textit{Threatened Species Protection Act 1995} (Tas) provides for the declaration of “critical
habitats” – areas of land defined on a map which the Director determines as habitat critical to the survival
of a listed taxon of flora or fauna. This determination happens after consultation with the Scientific
Advisory Committee (SAC)\textsuperscript{54} and the Director must give public notification of the area determined as a
critical habitat by reference to a map registered in the central plan office\textsuperscript{55} showing the boundaries, extent
and details of the area\textsuperscript{56} and notify any landholder or other person who is likely to be affected by the
determination.\textsuperscript{57} The Director must, within 30 days publish notice of the determination in the Gazette,\textsuperscript{58}
unless the Minister is of the opinion that disclosure of the location of the habitat would result in any
harm being done to it or to the flora or fauna which it supports\textsuperscript{59} and cannot publish the determination of
a critical habitat that is on private land unless the landholder agrees.\textsuperscript{60}

The Minister may make an interim protection order to conserve the critical habitat of a listed taxon of flora
or fauna or a nominated taxon of flora or fauna which has been accepted by SAC for listing and which in
either case is on private land or Crown land and not subject to a public authority agreement.\textsuperscript{61} In making

\begin{itemize}
\item \textsuperscript{46} \textit{Biodiversity Conservation Act 2016} (WA) s 56 (1).
\item \textsuperscript{47} \textit{Biodiversity Conservation Act 2016} (WA) s 57.
\item \textsuperscript{48} \textit{Biodiversity Conservation Act 2016} (WA) s 59 (1).
\item \textsuperscript{49} \textit{Biodiversity Conservation Act 2016} (WA) s 59 (4).
\item \textsuperscript{50} \textit{Biodiversity Conservation Act 2016} (WA) s 59 (5).
\item \textsuperscript{51} \textit{Biodiversity Conservation Act 2016} (WA) s 64.
\item \textsuperscript{52} \textit{Biodiversity Conservation Act 2016} (WA) s 65 (1).
\item \textsuperscript{53} \textit{Biodiversity Conservation Act 2016} (WA) s 67.
\item \textsuperscript{54} \textit{Threatened Species Protection Act 1995} (Tas) s 23 (1).
\item \textsuperscript{55} Under the \textit{Survey Co-ordination Act 1944} (Tas).
\item \textsuperscript{56} \textit{Threatened Species Protection Act 1995} (Tas) s 23(2)(a).
\item \textsuperscript{57} \textit{Threatened Species Protection Act 1995} (Tas) s 23(2)(b).
\item \textsuperscript{58} \textit{Threatened Species Protection Act 1995} (Tas) s 23(5).
\item \textsuperscript{59} \textit{Threatened Species Protection Act 1995} (Tas) s 23(6).
\item \textsuperscript{60} \textit{Threatened Species Protection Act 1995} (Tas) s 23(7).
\item \textsuperscript{61} \textit{Threatened Species Protection Act 1995} (Tas) s 32(1).
\end{itemize}
an interim protection order, the Minister must consider (1) matters relating to nature conservation; and
(2) the social and economic consequences of making the order; and (3) if the order relates to private
land, any comments made by Community Reference Committee (a committee made up of nine members
appointed by the Minister consisting of people representing local government, farming, rural, forestry,
fishing industries and the Scientific Advisory Committee); and (4) any other relevant matters.62 An interim
protection order ceases to be in force after (1) if the order relates to Crown land, a period of 65 business
days; or (2) if the order relates to private land, a period of 30 business days,63 but the Minister may, with
the agreement of all persons affected by an interim protection order, extend the period during which the
order is in force.64 Terms and conditions as are specified in the order and may provide for protection
and management of flora, fauna and the land within the habitat which is the subject of the order and the
prohibition or regulation of any activity which takes place on the land or the use and management of
the land within the habitat or activity which takes place outside the habitat which is the subject of the
order but which is likely to affect the habitat adversely.65 It is an offence not to comply with the order.66
No critical habitats have been listed and no interim protection orders have been declared in Tasmania.67

**DISCUSSION**

The importance of unburnt refuges, and their identification and management, has rightly featured
prominently in the initial responses from government agencies to the ecological impacts of the 2019–2020
bushfires. Considering the importance of refuges in the immediate recovery processes and the threats
these refuges may face from feral animals, weeds and potentially from a range of land management
activities (including logging, clearing, grazing, inappropriate fire regimes and other human-influenced
degradation processes), the application of critical habitat listing provides additional legal protection and
management focus.

Most State and Federal threatened species legislation in Australian jurisdictions have provision
for recognising critical habitat for threatened species. However, the process for how critical habitat
is identified, recognised, applied and enforced varies between jurisdictions, and the mechanism has
generally been considered to have been under-utilised for various reasons.68

For example, among a range of reasons for under-utilisation, the Australian Network of Environmental
Defender’s Offices Inc suggested ‘Reluctance to use the Critical Habitat Determination and Interim
Conservation Order (ICO) provisions of the [Victorian Flora and Fauna Guarantee Act 1988] could be
due in part to the requirement in the Act to pay compensation to landholders for financial loss suffered
as a direct and reasonable consequence of the making of an ICO.’69 At the federal level, the Australian
Conservation Foundation (ACF), a proponent for reforming the Critical Habitat listing process under
the **Environment Protection and Biodiversity Conservation Act 1999** (Cth) (EPBC Act), outlined an
eexample of where a lack of enforcement and prosecution powers under that Act for any land other than
Commonwealth land may be influencing (and inhibiting) critical habitat listing.70 It cites the case of

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62 Threatened Species Protection Act 1995 (Tas) s 32(3).
63 Threatened Species Protection Act 1995 (Tas) s 32(6).
64 Threatened Species Protection Act 1995 (Tas) s 32(8).
65 Threatened Species Protection Act 1995 (Tas) s 33.
66 Threatened Species Protection Act 1995 (Tas) s 36(3).
67 ANEDO, n 42.
69 ANEDO, n 42.
70 ACF, n 68.
the Critically Endangered Leadbeater’s Possum (*Gymnobelideus leadbeateri*) where the Draft National Recovery Plan for the species stated that “…all current and prospective suitable habitat is critical for its survival, and necessary for its recovery”. This would mean that suitable montane ash forest in Victoria’s Central Highlands, including areas available for timber harvesting, are critical for the survival of that species. ACF reported obtaining documents through Freedom of Information which “reveal, the TSSC [Threatened Species Scientific Committee] agreed at its September 2016 meeting to advise the Environment Minister that under Australia’s current tenure constrained critical habitat laws, ‘there would be no clear conservation benefit from pursuing a listing on the Register of Critical Habitat for this species’”. The *Guardian* reported a spokesperson for the Australian Government’s Department of the Environment and Energy as stating the reason for this was because habitat for the Leadbeater’s Possum was not found on Commonwealth land [quoting the departmental spokesperson]: “The range of the Leadbeater’s Possum does not include Commonwealth land. The committee agreed that placing any of its habitat on the register would therefore have no conservation benefit”.

The under-utilisation of critical habitat provisions in Australian legislation is in contrast with the US *Endangered Species Act of 1973*, where critical habitat designations have been used more extensively. There, reviews have found endangered species which had critical habitat listed were more likely to be stable or improving than species that had no critical habitat protection within two years of the listing. Beyond two years, species with critical habitat protections were twice as likely to be improving in terms of population size than those without protections.

While there has been reluctance at times by agencies to designate Critical Habitat in the United States, Owen found that “even if critical habitat does not substantially change the [US Fish and Wildlife] Services’ regulatory approaches, regulated entities seem to believe that designations do increase regulatory stringency, and that belief may also deter some activities that might otherwise harm species”. This is an important counter to reluctance to use critical habitat due to legal strength alone. Mapping and publicly identifying critical habitat would likely add extra political and/or reputation pressure for public agencies or private landholders that might willingly damage this habitat, regardless of an inability to prosecute, particularly considering significant public concern for populations of threatened flora and fauna species as a result of 2019–2020 bushfires.

Nonetheless, as the vast majority of nationally listed threatened species, and the habitat that is critical to their survival occur on land other than Commonwealth land, this is clearly a deficiency in the Federal *EPBC Act*. The second Independent Review of the *EPBC Act* is currently underway and is due to be completed later in 2020. The discussion paper produced for this review does not specifically mention

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72 ACF, n 68, 11–12. That report cites “Draft Minutes, 65th Meeting of the Threatened Species Scientific Committee, 6–8 September 2016” as the document secured under Freedom of Information.


77 That is, examples in ACF, n 68; Cox, n 73.

critical habitat provisions, but it does invite proposals for reform of the Act.79 Making critical habitat listing process more efficient and transparent, as well as having enforcement provisions across all land tenures, would be an obvious opportunity for reform, particularly for it to be flexible enough to be effectively deployed for emergency events such as 2019–2020 bushfires.

While State and Territory governments have a range of different mechanisms to protect biodiversity on public80 and private land,81 critical habitat listing has the ability to be “tenure-blind” and recognises the need to protect this important element of biodiversity regardless of where it is located.

In reforming Victoria’s Flora and Fauna Guarantee Act 1988 (Vic), there was wide support for “adopting a cooperative approach to managing critical habitat through agreements. Respondents to public consultation on that reform suggested an incentive framework was necessary to provide funding to support landowners significantly affected by a critical habitat determination.”82 Compensation for landholders are part of critical habitat provisions in a number of jurisdictions and should be considered for others, but these should also be considered in a broader need for reform to Australian taxation legislation to better encourage private land conservation.83 Critical habitat listing and associated interim conservation orders or equivalent (for jurisdictions that have them) may only need to be interim measures. Indeed, it may be that much unburnt refuges that might qualify as critical habitat post-bushfires only need to be listed as a temporary measure until (1) management efforts such as feral animal and weed control84 provide important on-ground recovery outcomes, (2) the surrounding forest regenerates (and habitat within that recovering forest increases) and (3) as the status of threatened species becomes clearer/improves.

Predications for more frequent and extreme weather events, including catastrophic bushfires associated with climate change,85 are likely to result in more events that result in temporary or permanent loss of habitat for threatened species. This means there will be an increasing need to apply, and where necessary, reform, critical habitat listing so that the aims of these provisions can be quickly enacted when needed most. Processes for identifying and listing critical habitat under legislation in various jurisdictions typically involve an assessment and recommendations of a scientific advisory committee or equivalent. If critical habitat listing is to be useful in urgent situations, such as the response to the 2019–2020 bushfires, this will require rapid mobilisation of these committees to assess likely critical habitat. Providing extra resources to the national Threatened Species Scientific Committee and equivalent State scientific committees to expedite the identification of critical habitat will be essential. Ideally, there would be co-ordination between committees in different jurisdictions, particularly considering the number of threatened species whose distribution cross jurisdictional boundaries, and deliberations would use consistent and/or best available data.

It is recognised that rapidly assessing critical habitat for a vast range of species over a large area is a considerable task. Considering the importance of unburnt refuges, applying the precautionary principle may mean than many or most unburnt areas within or adjoining fire scares would qualify as critical habitat without full surveys of these refuges to determine species occupancy or immediate suitability. A more structured approach to critical habitat listing, particularly following situations as experienced in

80 For example, B Coffey, JA Fitzsimons and R Gormly “Strategic Public Land Use Assessment and Planning in Victoria, Australia: Four Decades of Trailblazing But Where to From Here?” (2011) 28 Land Use Policy 306.
84 DELWP, n 5; DPIE, n 6.
these bushfires (large amounts of habitat burnt for a large number of threatened species and the need for quick action, with at times limited data)\textsuperscript{86} would bring increased certainty to species recovery efforts, to industry and to private landholders alike, and ensure recovery efforts are best spent where needed.

In the absence of clear process and direction, interim protection may occur in less planned processes, such as through the courts or political process. This is already playing out – the Australian Broadcasting Corporation reported in late January 2020 that the Supreme Court of Victoria:\textsuperscript{87}

\begin{quote}
[Gr]anted an interim injunction ... preventing logging in three coupes – areas to be harvested in the forests — ahead of a full hearing [in February 2020]. Justice Kate McMillan said there was “a real threat of a serious or irreversible damage to threatened species and their habitat should harvesting operations continue in the coupes”. “The recent bushfires have caused extensive environmental damage, the severity of which is only beginning to be understood,” Justice McMillan said.
\end{quote}

Similar concerns on plans for timber extraction in potentially important Koala habitats in New South Wales post-bushfires have been raised by the chair of NSW Upper House inquiry into koala populations and habitat.\textsuperscript{88}

Society, through inclusion of provisions for critical habitat designations in threatened species legislation in most Australian jurisdictions, value critical habitat and expect it to be protected. The bushfires of 2019–2020 has made the concept of critical habitat and the need to legally designate it using existing provisions more important than ever. However, the extensive nature of these bushfires and deficiencies in scope and process of critical habitat listing provisions in some jurisdictions highlight the need for reform of these provisions to ensure they are fit-for-purpose and able to respond effectively to future catastrophic events.

\textsuperscript{86} Acknowledging in such cases the tradeoffs between timeliness and the full consideration of critical habitat needs for a threatened species; see also AE Camaclang et al, “Current Practices in The Identification of Critical Habitat for Threatened Species” (2015) 29 Conservation Biology 482.
