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Hope and caution: Rewilding to mitigate the impacts of biological invasions

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Summary

Rewilding is a novel approach to ecological restoration. Trophic rewilding in particular aims to reinstate ecological functions, especially trophic interactions, through the introduction of animals. We consider the potential for trophic rewilding to address biological invasions. In this broad review, we note some of the important conceptual and ethical foundations of rewilding, including a focus on ecosystem function rather than composition, reliance on animal agency, and an appeal to an ethic of coexistence. Second, we use theory from invasion biology to highlight pathways by which rewilding might prevent or mitigate the impacts of an invasion, including increasing biotic resistance. Third, we use a series of case studies to illustrate how reintroductions can mitigate the impacts of invasions. These include reintroductions and positive management of carnivores and herbivores including European pine martens (*Martes martes*), Eurasian otters (*Lutra lutra*), dingoes (*Canis dingo*), Tasmanian devils (*Sarcophilus harrisii*) and tule elk (*Cervus canadensis nannodes*). Fourth, we consider the risk that rewilding may enable a biological invasion or aggravate its impacts. Lastly, we highlight lessons that rewilding science might take from invasion biology.

Introduction

Various versions of rewilding have been described since the term was first coined in the 1990s [1]. Recently, Svenning *et al.* [2, p898] defined trophic rewilding as “species introductions to restore top-down trophic interactions and associated trophic cascades to promote self-regulating biodiverse ecosystems”. This and similar approaches to rewilding focus on the introduction of vertebrate populations or their functional analogues to landscapes from which they have been extirpated. The primary aim is often to restore top-down trophic interactions, although this does not preclude restoration of other functions such as physical disturbance of substrate or dispersal of seeds. In contrast to much current environmental policy [3], rewilding is focussed on ecosystem functions rather than species and assemblages. Autonomy is a guiding principle – reintroduced populations should be self-sustaining – and the restorative effects are expected to be long term and large scale.

Rewilding typically involves reintroducing species to ecosystems that have been modified from their original state. In addition to habitat modifications, species loss, and climate change, many ecosystems across the globe have been severely impacted by biological invasions [4]. Invasions have been a major cause of extinctions and, in synergy with other factors, continue to threaten many species [5-7]. While rewilding has been undertaken or proposed with various goals in mind, few studies have considered rewilding in the context of biological invasions.

Our aim here is to review the potential for trophic rewilding to prevent, mitigate or enhance the impacts of biological invasions. We restrict our review to the effects of reintroducing large carnivores and herbivores, although rewilding programs could involve the introduction of other agents of change, such as pollinators, seed dispersers or ecosystem engineers. First, we note some of the important conceptual and ethical foundations of rewilding; second, we use theory from invasion biology to highlight two pathways by which rewilding might prevent or mitigate the impacts of an invasion; third, we illustrate these points through a series of case studies, and; fourth, we consider the risk that rewilding could enable invasions or aggravate their impacts.

Ethical and conceptual considerations

Conservation is arguably becoming more interventionist. Corlett [8] has identified three current paradigms that reflect this in different ways: conservation translocations, novel ecosystems, and rewilding. The latter interventions focus on ecosystem function, as opposed to composition [9], which is also reflected in recent calls for a 'new conservation' [10]. These paradigms sometimes distance themselves from traditional, preservationist conservation approaches, which are described, perhaps unfairly, as reactive and pessimistic [11, 12].

Two key concepts that support trophic rewilding are non-human agency and coexistence. A presumption of non-human agency is evident in the rewilding literature [e.g. 11, 12, 13, 14], especially in the preference for autonomy in the restored ecosystem, the deployment of animals to do restoration, calls for 'open-ended' conservation, tolerance for unexpected outcomes, and the adaptation of the word 'wild' which denotes, among other things, self-willed nature [15]. Various theorists have promoted non-human agency as a necessary feature of an adequate environmental ethic [e.g. 16, 17]. Having regard for agency may also help reconcile conflicting obligations to environmental integrity and the welfare of other creatures. This is a key problem for environmental ethics [18] and a particularly difficult problem when managing biological invasions [19].

Agency, in particular autonomous agency, can entail an ethic of respect [e.g. 20]. Something similar is at play in rewilding: Rewilding advocacy often includes an assumption that coexistence of disparate groups is possible and even preferable to exclusion, for example coexistence between humans and large predators [21-23]. An ethic based on coexistence may be a viable alternative to exclusion-based approaches to conservation such as the separation of large animals and humans [21], a focus on protected areas [24] or a wilderness-focussed ethic [25]. However, coexistence in this context requires more detailed explanation.

What implications does this new paradigm have for biological invasions? First, there may be moral implications. If coexistence were broadly accepted then managers would have greater warrant to reintroduce large animals. It could lead to wider acceptance that preservation is not a default conservation aim, releasing managers from an *a priori* obligation to return the system to some pristine state. Similarly, the mere presence of a non-native species need not be considered a failure of management. Practically speaking, managers often have insufficient resources to eradicate non-native species anyway [26] so this might seem to make little practical difference. However, thinking about ecosystems and animals in terms of function, agency, and coexistence might inspire novel solutions.

Biological control programs are founded on a similar approach and involve the introduction of organisms with the purpose of altering invasive populations. Conceptually at least, trophic rewilding differs from classical biological control in several ways [27]. It focusses on large vertebrates rather than including invertebrates; values reintroduced populations for their own sake rather than as means to an end; concentrates on non-commercial ecosystems; aims for influence on many species rather than a single species; and preferences species assumed to have coevolved with the community, reducing the risk of negative influences (but see below). Beyond biological control, examples of agent- and function-oriented approaches include recent proposals to introduce Tasmanian devils to mainland Australia to regulate cat and fox impacts on small mammals (described below); behavioural and evolutionary modifications to 'train' species to avoid novel predators; and targeting individual predators that cause the most damage to vulnerable populations [28-30].

The discipline of invasion biology is now well established [7]. It is predicated on the fact that some populations that are new to a place cause problems, including undermining important ecological functions and threatening species with extinction. However, some concepts and language commonly associated with invasion biology have been recently criticised, notably the militaristic and xenophobic framing of problems,

the seeming incommensurability with animal welfare concerns, and the assumption that any influence of a new population is negative [31-33]. It is increasingly recognised that introduced or newly arrived populations can be valuable additions to an ecosystem, a view which has obvious overlap with rewilding advocacy [33, 34]. Recognising the above issues, we apply the term 'invasive' only to those new arrivals that cause significant negative impacts.

Invasion processes

Invasion biology has generated several prominent hypotheses to explain invasion processes [35]. However, theoretical generalities explain far less variation than we would like and outcomes are strongly affected by local context, unpredictable ecological dynamics, and a multiplicity of processes [36]. This precludes general statements about whether rewilding will work to prevent or mitigate biological invasions.

On the other hand, rewilding might work against invasions in specific circumstances. There are several biotic processes that constrain invasions [37]. Rewilding can increase biotic resistance, i.e. reduce the probability of successful establishment or spread of an invasive population. Invasions are often described in terms of stages, e.g. arrival, establishment and spread, and the barriers between them [38]. Biotic interactions play an important role in strengthening barriers at each stage [35, 37], though it is doubtful whether communities can completely repel newcomers [37].

More specifically, once an invasive population is established, a rewilding introduction might contain, prevent or reduce impacts, for example via top-down trophic processes that suppress the invasive species and promote its competitors. Carnivores can suppress invasive populations directly by predation. Herbivores might selectively feed on exotic species, reducing their abundance; or by feeding more broadly prevent any one species (including an invasive species) from dominating. We present several case studies below, focussed on rewilding with carnivores and herbivores.

Rewilding with carnivores

Trophic rewilding proposals have focussed on large mammals [2]. The ecological influence of large predators and their disproportionate vulnerability to extinction have heightened their importance in conservation [39, 40]. However, we are aware of no cases where a large terrestrial predator has been introduced to its former historic range to control a biological invasion and to persist as a valued part of the assemblage.

Top predators can have strong influences on ecosystems [39, 40], primarily mediated through trophic cascades and resource facilitation [40]. For example, large body size in mammalian predators is associated with a narrower prey base. Due to energy and intake constraints, most species larger than 21.5 kg feed on vertebrate prey only, typically species similar to their own body mass or larger [41]. For most taxonomic groups there are fewer large-bodied species [42]. Thus, large predators targeting large prey tend to directly impact few species.

Despite this specificity, cascading impacts can be dramatic. For example, the reintroduction of wolves (*Canis lupus*) into Yellowstone National Park in 1995-6, following an absence of more than seven decades, resulted in changes to Rocky Mountain elk (*Cervus elaphus*) abundance and spatial habitat use. This in turn may have influenced woody vegetation recruitment, species composition and biomass, along with riparian songbird and beaver populations [43], although there is uncertainty about the strength of the influence of wolves in this landscape relative to other factors, such as drought and the human hunting of elk [44-46].

Predators can dampen variability in prey abundance and may increase community resilience to climate change [47]. Predators can provide carcasses for scavengers. They are often also scavengers themselves and influence other species via resource-mediated competition [48]. These processes point to the potential for top predators, especially large carnivores, to keep invasions in check via top-down trophic pressure.

Carnivores can also prevent the establishment and spread of invasive populations. Carnivore guilds are often characterised by strong, aggressive intraguild interactions, including harassment, kleptoparasitism, and intra-guild killing without consumption [49-51]. Like prey species, other predators often respond to the presence of a predator with spatial and temporal avoidance [51]. When the disrupted population is invasive, these processes represent modes of biotic resistance, i.e. preventing the establishment of new populations or their spread into new areas.

Case 1: Martens and squirrels in the British Isles

Following introductions around the turn of the twentieth century, grey squirrels (*Sciurus carolinensis*) have displaced red squirrels (*Sciurus vulgaris*) in Britain and elsewhere, through exploitative competition. This process has been aggravated by squirrelpox virus, a novel pathogen possibly introduced with the grey squirrels, which is lethal to red squirrels but asymptomatic in grey squirrels, resulting in disease-mediated apparent competition [52].

European pine martens (*Martes martes*) are generalist predators native to most of Europe [53, 54]. Because of persecution and habitat loss, they are now rare in the British Isles and functionally extinct in some areas [55]. Populations in Scotland have recently increased following new legal protections, reforestation and conservation introductions [54].

Sheehy *et al.* [56] have shown that the recovery of the pine marten in Britain has favoured red squirrels over grey squirrels where pine marten density is sufficiently high. Predation by pine martens probably benefits red squirrels via predator-mediated apparent competition and pathogen dynamics. Predatory impacts of pine martens are much greater on grey than red squirrels [57], reducing pathogen spillover of squirrelpox from its reservoir in grey squirrels. Modelling by Sheehy *et al.* [56] predicted near-zero probability of grey squirrel presence in landscapes with high pine marten connectivity, indicating that predation by pine martens may be able to severely suppress grey squirrel populations and promote coexistence with red squirrels.

Case 2: Mink and otters in Europe and South America

American mink (*Neovison vison*) were introduced to Europe, the former USSR, and Tierra del Fuego for fur farming, beginning in the 1920s. Subsequently, escaped animals established wild populations [58-60], which are now widely distributed across much of the European continent and continue to spread across the Tierra del Fuego archipelago [61]. American mink threaten a wide range of predator and prey species in Europe, including the critically endangered European mink (*Mustela lutreola*), the European polecat (*Mustela putorius*), voles, shrews, birds, frogs and fish [59, 62-67]. In Patagonia, they prey particularly on native rodents but also on a wide variety of ground-nesting birds and fish [68, 69].

Abundance of Eurasian otters (*Lutra lutra*) declined sharply in Britain in the middle of the twentieth century due to hunting and insecticide pollution [70]. Eurasian otters are larger than American mink, they are better swimmers, and there is evidence that they outcompete American mink for food [71].

Following the experimental release of otters into an area only occupied by mink, Bonesi and Macdonald [72] observed a reduction in the proportion of sites occupied by mink and a reduction in mink abundance, range, and range integrity compared to control sites with no otter release. The authors attributed the effect to aggressive exclusion and predation by otters, i.e. interference competition. This pattern is reflected in Patagonia, where southern river otters (*Lontra provocax*) are unaffected by the presence of mink, can suppress their abundance and temporal activity, force dietary and habitat shifts, and potentially prevent their occupancy in sympatric areas [60, 73].

Case 3: Top predators in Australia

Australia has lost 29 endemic land mammal species since European colonisation, the highest rate of extinction of mammals of any continent over the last 200 years [74]. A primary cause has been predation by two introduced mesopredators: red foxes (*Vulpes vulpes*) and feral cats (*Felis catus*) [74, 75]. Predation pressure has been enabled and enhanced by factors such as habitat destruction, the loss of traditional burning regimes, and widespread suppression of the apex predator: the dingo (*Canis dingo*) [76, 77].

Persecution of dingoes, intended to prevent attacks on livestock, accounts for their low abundance or absence across much of their former range [78]. There is evidence that dingoes regulate mesopredators, benefitting small vertebrates threatened by cats and foxes [79, 80]. Dingoes also prey on macropods and other large herbivores. Predator-control of these herbivores can result in more vegetation of greater structural complexity [50, 80, 81] which can in turn help small mammals to avoid predation [82].

The strengths of these effects may be dependent on assemblage and bioclimatic region [83] and productivity may play a significant role [84]. Several authors have proposed positive management – relaxation of lethal control of dingoes and even reintroduction – to reduce extinction risk for native fauna through suppression of cats and foxes [85]. This is unlikely to gain community support in the sheep rangelands of Australia without significant shifts in attitudes toward predators.

More speculative is the suggestion that the Tasmanian devil ('devil', *Sarcophilus harrisii*) could be introduced to the mainland of Australia to reduce the impacts of cats and foxes on threatened vertebrates [86]. Dingoes arrived in Australia at least 3500 years before present [87]. Both the thylacine (*Thylacinus cynocephalus*) and the devil were widespread on the mainland of Australia at the time, and persisted until about 3200 years before present [88]. Several factors have been implicated in the mainland extinction of the devil and the thylacine. These include climate, humans and dingoes, although recent modelling suggests that the influence of dingoes was not strong [89, 90]. Dingoes never reached Tasmania and the thylacine was hunted to extinction there in the twentieth century, leaving the devil as the largest predator [91].

There is evidence that devils can influence the abundance and activity of cats, with positive effects on small and medium sized mammal populations [92-96, cf. 97]. In a long term, wide ranging study, Hollings *et al.* [94] found that the strength of top-down suppression of cats varied by region. Devils appeared able to suppress cat activity and/or abundance in forested areas with few human settlements and higher rainfall. However, in drier agricultural areas this top-down effect was weaker. Factors related to food resources were more important for predicting cat occurrence. These factors included rainfall and the availability of prey, particularly the European rabbit (*Oryctolagus cuniculus*), a key prey species for cats in agricultural areas. Rabbit abundance was likely influenced by rainfall and outbreaks of rabbit haemorrhagic disease. Cats in Tasmania may be suppressing populations of the endangered eastern quoll (*Dasyurus viverrinus*), preventing their recovery by preying upon juveniles (Figure 1) [95].

The reintroduction of devils to the mainland of Australia may be a viable option to promote the coexistence of cats, foxes and threatened prey species, complementing the influence of dingoes. Modelling based on body size and diet suggests that devils could suppress the abundance and activity of cats, foxes and wallabies, benefitting small and medium sized mammals and their habitat [86, 98, cf. 99]. The same modelling suggests that dingoes and devils could coexist. Species distribution modelling points to available habitat in south-eastern Australia, including areas where dingoes are scarce [86].

Rewilding with herbivores

Large herbivores influence ecosystems by removing plant biomass, changing vegetation structure, increasing light at ground level, moving large amounts of soil, dispersing nutrients, reducing fire-fuel loads and dispersing seeds over long distances [100]. These influences are often strong for the largest-bodied species [e.g. 101, 102, 103]. Some influences are species-specific, for example, hundreds of plants have propagule dispersal syndromes that reflect a strong mutualistic association with megaherbivores [104]. Others are more generalised, for example, larger bodied herbivores tend to consume a broader range of structural plant material than smaller herbivores and are generally less selective among different plant species on offer [105, 106] so they have large effects on plant biomass and vegetation density. These processes are potential means of increasing biotic resistance or applying top down trophic pressure and could inspire a range of imaginative rewilding solutions to invasion problems. We focus on the most direct means, the consumption of plant matter, and highlight two mechanisms by which introduced herbivores could reduce the abundance of invasive plant populations: by preferentially grazing invasive plants relative to native plant species, and by removing the most abundant species and preventing dominance by any one species.

Herbivory can prevent the establishment of exotic plants and reduce their performance under a range of conditions [37]. This seems to be driven in part by the biogeographical context of the plant and the consumer. A meta-analysis of manipulative field studies of mainly vertebrate herbivory found that the relative abundance of exotic plants tended to be decreased by native herbivores but was increased by exotic herbivores [107]. This pattern held for exotic plants further classified as invasive.

The above suggests that the reintroduction of a vertebrate herbivore, or the introduction of functional replacement of an extinct species, could reduce the abundance of an invasive population and mitigate its impacts through preferential feeding, that is, promote its benign coexistence in a landscape. A well-studied

example is the introduction of two species of tortoise to Round Island, near Madagascar [108]. Following the eradication of goats and rabbits, the loss of grazing pressure resulted in invasive exotic plants and some fast-growing native plants dominating grasslands. Mauritian giant tortoises (*Cylindraspis inepta* and *C. triserrata*) were once grazers on the island but are extinct. Populations of two species unknown from the island, Aldabran giant (*Aldabrachelys gigantea*) and Madagascan radiated (*Astrochelys radiata*) tortoises, were introduced. These species browsed primarily on exotic plants, along with a few native species that became abundant after the eradications. In this case, coevolution of the herbivores and plants was evidently crucial. Species that coevolved with giant tortoises have traits that reduce browsing pressure relative to fast growing yet more palatable invasive plants on Round Island

For some systems, a range of large generalist herbivores might perform a similar function and biogeographic history may be less important. In these systems, grazing by exotic generalists might prevent exotic plants from dominating. For example, grazing by cattle was found to be effective at maintaining plant diversity on vernal pools in California, substantially reducing the cover of exotic annual species and increasing the cover and richness of native plant species [109]. However, the influence of herbivory is not always so straightforward.

Case 4: Tule elk and velvet grass in California

Tule elk (*Cervus canadensis nannodes*) is a subspecies endemic to California. Hunting and land conversion in the nineteenth century drove numbers from the hundreds of thousands to a mere ten individuals. Legal protections allowed the herd to recover into the thousands through the twentieth century. Tule elk were introduced to several reserves, including at Tomales Point in 1978, a coastal grassland north of San Francisco. The site had been grazed by cattle for a century beforehand but cattle were removed as the elk were introduced. By 2003 the elk population in the 1,030 hectare Tomales Point Elk Reserve was approximately 500 individuals. Tule elk primarily consume herbaceous forbs and grasses but feed on shrubs during winter months [110].

In a five-year exclusion experiment, Johnson and Cushman [110] showed that within functional groups, for the most part, tule elk affected native and exotic plants similarly. Herbivory reduced the biomass of both exotic and native perennials and increased the biomass and abundance of both exotic and native annuals [110]. These effects are of potential benefit to Californian grasslands that have been invaded by perennial shrubs [111].

Velvet grass (*Holcus lanatus*) is widely invasive in Californian grasslands. In Tomales Point Elk Reserve, elk reduced both the abundance and biomass of velvet grass, probably through herbivory and trampling. However, velvet grass escaped herbivory when associated with a native shrub, *Baccharis pilularis* [110]. Subsequent work demonstrated that tule elk were less effective in reducing velvet grass biomass in *Baccharis*-dominated grasslands. Soil heterogeneity (in terms of pH and moisture) was a stronger influence than herbivory on velvet grass success. Elk did not prevent the spread of the species to new areas [112]. This parallels outcomes in Californian grasslands, many of which are highly invaded, under managed livestock grazing. A meta-analysis of fifteen studies of Californian grasslands found that grazing consistently increased the cover of exotic forbs but that other responses by native and exotic species depended on factors such as precipitation, the seasonality of grazing and community type [113].

Causes for concern: making invasions worse

A critical lesson from studies of biological invasions is that the addition or removal of species from ecosystems can generate trophic cascades with unanticipated and sometimes unwanted consequences [114, 115]. As outlined above, community dynamics are notoriously unpredictable. Moreover, there are several cases where intentionally introduced species have become invasive, with widespread negative consequences [116, 117]. We highlight two important pathways by which trophic rewilding could aggravate invasion problems.

First, reintroduced or replacement species could themselves become overabundant. Despite having an ecological and evolutionary history in the recipient ecosystem, an introduced population may display novel properties: regulatory factors such as diseases or predators may no longer be operating; other novel species

might support larger populations; habitat or resources may have been altered to favour the introduced populations; or slight differences between an extinct species and its analogue might interact with these factors to induce a large impact. For example, the koala (*Phascolarctos cinereus*) is a tree-dwelling folivore with narrow dietary preferences. Several introduced koala populations, on islands and in isolated habitats on the mainland, have become so abundant that they have defoliated trees, resulting in local koala population crashes [118]. In 1972 at Sandy Point, Victoria, twenty individuals were introduced to a reserve and began feeding on manna gums (*Eucalyptus viminalis*). By the mid-1980s the koalas had severely defoliated many trees and 1,100 koalas were removed from a 200 ha area. Removals proved inadequate and by 1988 most of the remaining koalas had starved to death, having killed almost all the manna gums in that area [119].

Second, introduced species can facilitate other invasive populations [120]. For example, large populations of introduced rabbits and house mice (*Mus musculus*) in Australia can subsidise cat and fox populations, increasing predation pressure on declining species (figure 1) [121]. Effects can be indirect: Pigs (*Sus scrofa*) were introduced to the Channel Islands, USA in the nineteenth century. In the 1990s golden eagles (*Aquila chrysaetos*) colonised the islands, preying on the pigs. The pigs constituted a resource subsidy for the eagles and as a result, eagle predation almost caused extinction of an endemic fox (*Urocyon littoralis*). In addition, the abundance of skunks (*Spilogale gracilis*) increased dramatically because of release from competition with foxes [122].

Conclusion

Trophic rewilding can work to prevent biological invasions, mitigate their impacts and promote the coexistence of newcomer species with long time residents. However, the conditions for success will vary from case to case. Several processes may be at play, reflected in the multiple hypotheses generated to explain invasions. Local conditions will determine which processes pertain and to what degree. This carries two implications. Firstly, that general trends may have limited application to local problems. Secondly, that local conditions and proximate causes must be understood in detail before predictions can be made about rewilding and invasions.

Predicting the outcomes of biological invasions has proven difficult [123]. Predicting the outcomes of rewilding introductions is likely to be just as demanding [124]. If scientists engaged in rewilding are to avoid criticism on this point then quantitative methods of prediction are sorely needed. Related conservation practices such as *ex situ* conservation and managed relocations will also need these tools. Developing predictive methods will require a multidisciplinary approach, encompassing modelling [125], life history studies [126], long-term and landscape-scale ecological experiments [127, 128], palaeoecological investigation [129], climate and species distribution forecasting [128] and field trials [130] as well as input from invasion biologists.

Good predictive ability will not be enough to support decisions about rewilding. Rewilding carries risks and opportunities that range across social, legal, ethical, ecological and other domains. Many of these risks and opportunities are shared with conservation strategies such as conservation translocation and *ex situ* conservation [131]. Decisions about rewilding, and the development of policy and law to accommodate it, will require reflective discussion between stakeholders. It will also require the clear articulation of ethical principles [132, 133].

The concepts of agency, autonomy and coexistence hold potential for guiding both rewilding and invasion biology. However, they will not generate clear and agreed rules for conduct without further theoretical work and discussion. In the meantime, they represent starting points for conversation, reflection and research, and new ground from which to view environmental problems.

Finally, there are several lessons that scientists working on rewilding problems might take from invasion biology. Some research in invasion biology is directly applicable to rewilding, for example, work on the likelihood of newly arrived species to replace the functions of missing residents, or the importance of mutualisms for establishment success [134]. Aslan *et al.* [130] point out that risk management procedures for the use of biological control agents could be brought to bear on rewilding proposals. More broadly, there have been several recent attempts to create a unified theoretical framework for invasion biology [38], a difficult task considering the variety of phenomena described as biological invasions. Likewise for rewilding, the plurality of rewilding practices will likely preclude a simple and unified research framework. On the other hand,

rewilding will continue to encourage reflection, discussion, and imaginative solutions to biological invasion problems.

Figure and table captions

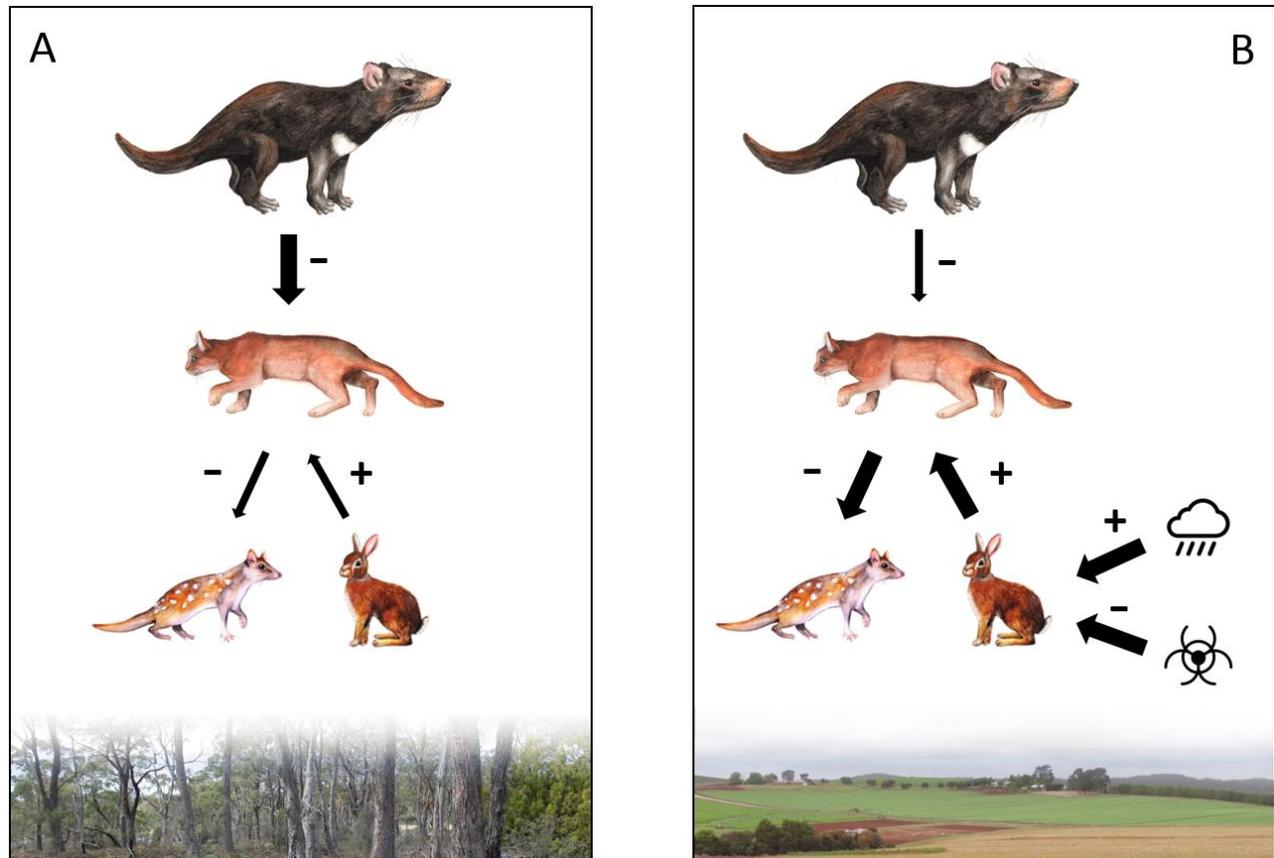


Figure 1. In Tasmania, the strength of top-down control by the apex predator was found to be mediated by productivity and climate: Devils suppressed cat activity and/or abundance in wet, forested areas (A). In drier, agricultural areas with more human settlements (B), bottom-up effects, including prey availability, were more important. High cat abundance, driven by rabbit abundance, may be preventing the recovery of the threatened eastern quoll [94, 95].

Additional Information

Information on the following should be included whenever relevant.

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Authors' Contributions

All the authors contributed to content, drafting and critical evaluation of the manuscript.

Competing Interests

We have no competing interests.

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