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PERSPECTIVES AND PARADIGMS

Integrating across knowledge systems to drive action on chronic biological invasions

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Abstract Designing and implementing long-term management strategies for chronic biological invasions is amongst the most vexing ecological research problems. Two key challenges to resolving this problem are: (a) integrating science-based and values-based (e.g. spiritual, cultural, economic and ethical) knowledge sources and (b) developing durable knowledge generation and curation platforms to co-ordinate long-term research efforts. We begin by identifying knowledge sources (stakeholder values, forecasts of invader spread and impacts, management technologies and operational logistics) to guide the high-level actions (governance framework design, selection of ethical management technologies, definition of long-term objectives, design of management strategies and operational plans implementing strategies) required for management of chronic invasions. We use exotic conifer invasions in New Zealand as an example. Next, we propose a transdisciplinary knowledge ecology framework where each knowledge source is represented by a separate knowledge generation and curation platform (i.e. knowledge ecosystem) and linked through high-level actions. We detail the structure and function of a single knowledge

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ecosystem (forecasting spread and impacts), and document two case studies to illustrate how knowledge ecosystems might (a) increase participation of individual researchers within long-term research efforts, and (b) facilitate inclusion of non-researchers in developing a common knowledge base. Finally, we propose a set of guidelines for combining science-based and values-based reasoning in decision-making via localised governance structures. We suggest that viewing knowledge ecologies as an integrated collection of distinct knowledge ecosystems offers a promising approach for identifying, generating, curating and integrating the knowledge sources needed to improve management of chronic biological invasions.

Keywords Conservation biology · Ecological forecasting · Ecosystem legacy effects · Knowledge integration · Large-scale eradication · Social-ecological systems

Introduction

Chronic plant invasions are among the most vexing environmental challenges, since eradication is often not practicable and containment is rarely possible (Hulme 2006; McGregor et al. 2012; Panetta and Cacho 2014). Minimising the impacts of chronic invaders requires sustained research effort spanning multiple biophysical, social and economic research disciplines (Hulme 2020), and often involves diverse stakeholders who may have conflicting objectives or values (Dickie et al. 2014; Head 2017; Kirk 2019).

The need to integrate research on complex environmental problems, such as biological invasions, across multiple organisations, science disciplines and agents (researchers, decision-makers and stakeholders) requires researchers to rethink how they organise and curate scientific knowledge towards addressing complex environmental problems (Dietze et al. 2018; Wallace et al. 2020). Further, the need to balance multiple stakeholder values when managing biological invasions requires approaches which transcend discipline-specific modes of enquiry (i.e. transdisciplinary research, Díaz et al. 2018; Groß and Stauffacher 2014; Lynch et al. 2015).We provide a synthesis of progress towards dealing with these dual challenges within a national effort to remove or contain non-native invasive conifers (MPI 2014).

Study outline

We begin by identifying the main sources of knowledge and high-level actions required to manage chronic biological invasions, focussing specifically on invasive conifers in New Zealand. We then propose a transdisciplinary knowledge ecology (Fam and Sofoulis 2017) structure composed of distinct knowledge generation and curation platforms (i.e. knowledge ecosystems Järvi et al. 2018) for each source of knowledge. Next, we describe a single section of the overall knowledge structure—forecasting invasive spread and ecosystem impacts—in detail and document its use in two case studies. The first shows how dynamic demography-based models of invasive spread permit rapid translation of new research to updated forecasts. The second documents a stakeholder engagement process (iterative focus groups, Nyumba et al. 2018) demonstrating how all stakeholders can improve both fitness-for-purpose and robustness of forecasting models. Finally, we use available literature to develop a set of guidelines for designing governance structures to integrate science-based and values-based knowledge in management decision-making.

Knowledge needs for chronic biological invasions as illustrated by New Zealand Wilding Conifer Invasions

Invasion by non-native Pinaceae, hereafter, 'wilding conifers', is well characterised internationally but the extent of and management responses to these invasions varies widely among systems (Nuñez et al. 2017; Richardson et al. 1994, 2014; Simberloff et al. 2010; van Wilgen and Richardson 2012). In New Zealand, the area occupied by naturalised conifer species has increased exponentially over the last few decades (Howell 2016), and these invaders occur on ca. 10% of the land area, albeit at low abundance (< 1 tree ha$-^{1}$) in most areas.[1] The national scale of wilding conifer invasion and management means a diverse range of

[1] Wilding conifer information system (WCIS) accessed February 2019. https://wildpines.linz.govt.nz/portal/apps/MapSeries/index.html?appid=107dc24b6a784d2a810ee664fa0a3036

stakeholders are involved, and that invasions occur across a wide range of environmental, ecological and landscape contexts, resulting in contrasting management challenges (MPI 2014). Achieving consensus on long-term goals, strategies, and research needs has been challenging for two reasons. First, deliberately established conifer plantations dominate New Zealand's forestry industry, which is a major land use occurring on about 10% of the land area (MPI 2014). Second, wilding conifers outside of plantations can have both positive impacts (e.g. reduced soil erosion, increased biomass carbon storage, and habitat for obligate forest species, Pawson et al. 2008) and negative impacts (e.g. reduced native plant diversity, soil carbon loss, reduced water yield, Mason et al. 2016). In turn, this generates different views or values regarding the impacts that conifers have on biological, economic or social values (Kirk 2019), especially when viewed from either indigenous (Māori) and non-indigenous (primarily Pākehā—New Zealand European) perspective (Harmsworth and Awatere 2013). These challenges may be common to management of many chronic biological invasions. Consequently, the types of knowledge, and modes of knowledge application, needed to manage wilding conifer invasions in New Zealand may also be common to many chronic biological invasions.

We propose five major knowledge sources required for management of New Zealand's wilding conifer invasions in particular and chronic biological invasions in general: Pākehā (non-indigenous) values, Māori (indigenous) values, spread and impacts forecasting, management technologies and operational logistics. We also identify five types of collaborative action, essentially high-level outcomes, integrating different knowledge sources, and focussed on the following high-level goals: bi-cultural governance structures, long-term management objectives, ethical management technologies, long-term strategies for achieving objectives and operational plans for implementing long-term strategies (Figure 1).

Explicitly incorporating invasive species impacts on stakeholder values (Head 2017) in decision-making is vital in securing stakeholder support for management objectives, setting policy and assessing competing management strategies (Graham et al. 2019). There is a growing understanding that successful management of complex environmental problems often requires governance (or other decision-making) frameworks which include all relevant stakeholders, constructively manage diverse stakeholder priorities and are mandated to make decisions over appropriate spatial and temporal scales (Bodin 2017). As a consequence of the enlightenment and colonialism, western rationalist value systems have historically been privileged in academia (Smith 2012) and environmental management (Harmsworth and Awatere 2013). However, recent high-profile science publications have emphasised the importance of including indigenous perspectives in environmental management (e.g. Díaz et al. 2018). Consequently, the need to incorporate of indigenous perspectives in management decisions may be common to many biological invasions.

In New Zealand, both government (informed primarily by Pākehā non-indigenous values) and indigenous Māori have a legal right to influence resource management planning and policy, but there are multiple barriers limiting this approach in practice (Harmsworth and Awatere 2013). The national wilding conifer management strategy (MPI 2014) makes no explicit provisions for Māori groups to influence management decisions. Indeed, Māori are not included in the list of "Key Participants in Wilding Conifer Management Governance" within the strategy.

Spatially-explicit invasive spread and impact forecasts for chronic biological invasions are vital in defining long-term management objectives which conform to stakeholder values (since the relevance of invasive species for stakeholder values is via their impacts, Head 2017). Spatially explicit forecasts also permit counterfactual comparison of outcomes from competing strategies (Caplat et al. 2014). Wilding conifer invasive spread and ecosystem impacts have largely been studied in isolation of each other in New Zealand (e.g. Buckley et al. 2005; Caplat et al. 2012; Dickie et al. 2011; Mason et al. 2016). This has hampered the incorporation of ecosystem impacts in prioritisation of management effort (Lloyd et al. 2016), which in-turn makes it very difficult to assess whether these priorities reflect stakeholder values.

Efficient detection (e.g. Martinez et al. 2020; Morisette et al. 2020) and ethical control technologies (Warburton and Norton 2009) are vital components in the management of biological invasions, particularly with regard to development of management strategies. Managing wilding conifer invasions poses significant

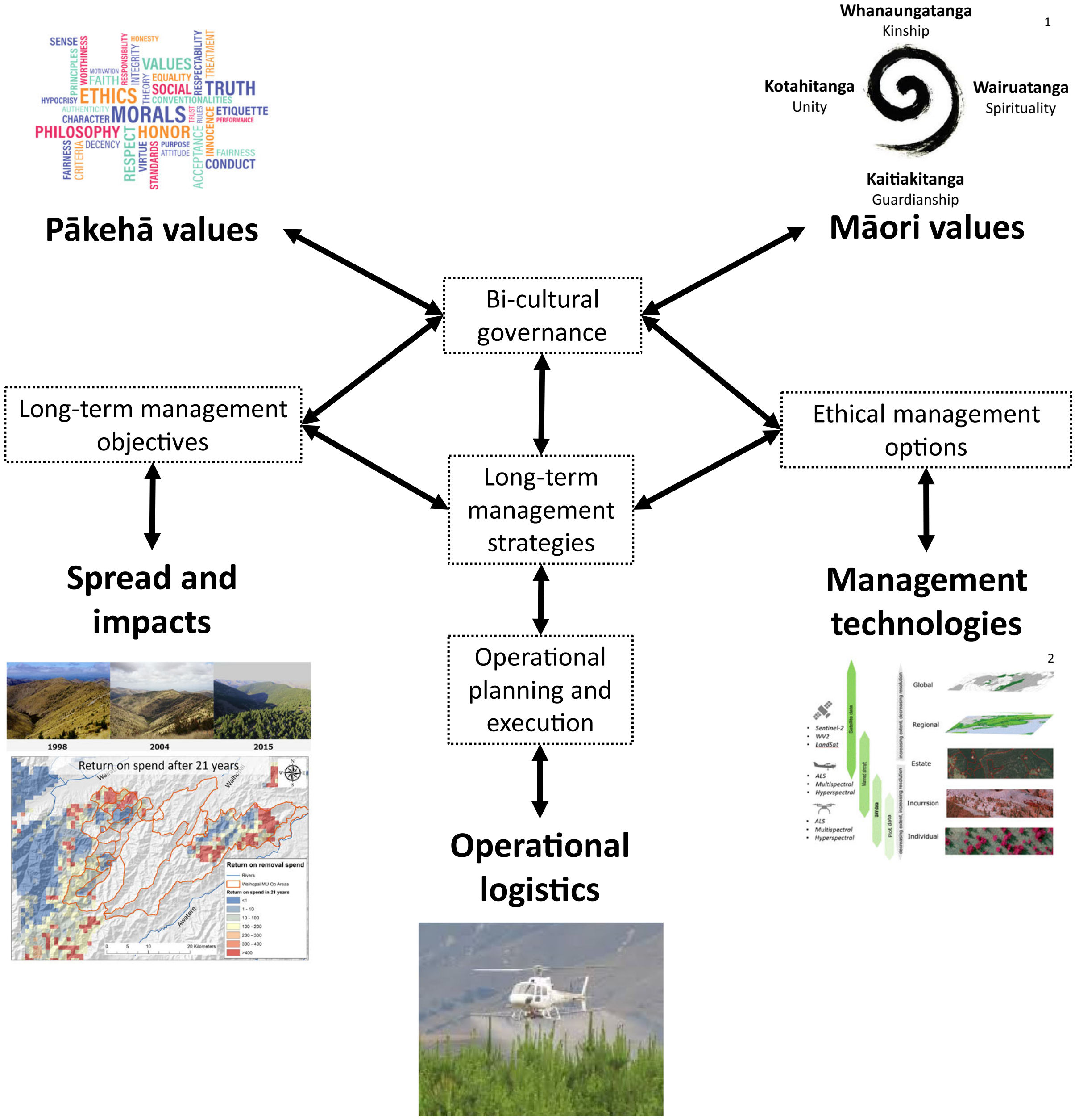

Fig. 1 A knowledge ecology structure for biological invasions in New Zealand composed of individual knowledge ecosystems in distinct research domains: Pākehā (non-Māori) values: social, ethical, economic, governance and policy research within a Pākehā world view; Māori (indigenous) values: social, ethical, economic and policy research within a Māori world view, employing indigenous (kaupapa Māori) research methods; Management technologies: scientific research on control and detection technologies; Spread and impacts: scientific research on conifer demography and ecosystem impacts; Operational logistics: scientific, social and economic research on the safe, efficient and ethical deployment of management technologies. The boxes with dashed borders represent the key collaborative actions via which individual knowledge ecosystems communicate: The double-headed arrows symbolise two-way knowledge exchange between knowledge ecosystems via collaborative actions. [1]Adapted from Henare (1998). [2]Adapted from Dash et al. (2017)

technological challenges in both areas. Conifer invasions span a range of density classes, from isolated individuals to dense forests. For scattered infestations, control is usually achieved via spot-spraying from a helicopter (Gous et al. 2015), which requires efficient detection of individuals before control operations to reduce search effort (Dash et al. 2017). For dense forest invasions (> 80% cover), control is achieved via blanket spraying from a helicopter, which requires herbicide formulations and delivery methods that provide acceptable kill rates without endangering the health of operators or the environment (Gous et al. 2014). Moderate density stands (20–80% cover) are particularly challenging since they often contain considerable indigenous plant biodiversity, which rules out blanket aerial herbicide application. However, for such infestations spot-spraying from a helicopter is prohibitively expensive (Gous et al. 2015) and ground-based control techniques are highly labour intensive, especially in difficult terrain. Consequently, the development of new detection and control technologies is a requirement for wilding conifer management.

Research on operational logistics (sensu Wallace et al. 2020) appears to have been rather neglected in the invasive species literature. Indeed, a web of knowledge search[2] with “operational logistics” and “invasive species” as search terms yielded a single result (Wallace et al. 2020, who use the term but do not link it to any cited literature). Searches with various combinations of related terms were similarly fruitless. This appears to be an important oversight, as the implementation of any management strategy is wholly dependent on effective operational deployment of management technologies. Publications which examine operational logistics (without using the term) in invasive species control programmes reveal that many unanticipated contingencies may hinder the safe, ethical and cost-effective deployment of management technologies in control operations (Head et al. 2015; Springer 2016). The national wilding conifer control programme produces safety guidelines for control operations (MPI 2019). However, we are not aware of any published research documenting the cost, effectiveness and safety outcomes of wilding conifer outcomes in operational (c.f. experimental or field trial) settings. This has potentially serious consequences for the accuracy of counterfactual analyses comparing the cost-effectiveness of competing strategies.

Using knowledge ecologies and knowledge ecosystems to implement transdisciplinary research principles

A major challenge to the engagement of individual biological invasions researchers in transdisciplinary research is the persistent debate over what is and what is not transdisciplinary research. This debate has been described as a “war of definitions” (Nicolescu 2010).

A concise definition of transdisciplinary research is the set of three axioms proposed by Basarab Nicolescu (text quoted from Nicolescu 2010):

The ontological axiom: “There are…in our knowledge of nature and society, different levels of Reality….”

The logical axiom: “The passage from one level of Reality to another is ensured by the logic of the included middle.”

The complexity axiom: “…perception [of Reality] is a complex structure: …all the levels [of Reality] exist at the same time.”

The ontological axiom treats knowledge gained through different ways of “knowing” the object of research (e.g. knowledge relating to biological invasions held within different disciplines) as representing different levels of reality. The logical axiom requires us to find novel techniques to navigate the logical space between disciplinary boundaries. The complexity axiom simply requires us to accept plurality: (i.e. that different perspectives on a given research object are not mutually exclusive).

For transdisciplinary research, we need a knowledge structure that permits non-pluralistic expression of different levels of reality (i.e. discipline-specific knowledge frameworks), accepts plurality of knowledge held in different disciplines that is relevant to the challenge we are addressing (i.e. biological invasions) and explicitly deals with the logical space between disciplines (Nicolescu 2010). The related concepts of knowledge ecologies (Fam and Sofoulis 2017) and knowledge ecosystems (Järvi et al. 2018), when applied together, provide a knowledge structure that achieves this.

[2] Accessed 25th of May 2020.

Both knowledge ecologies and knowledge ecosystems link different types of "knowers" (or agents—either individuals or institutions Fam and Sofoulis 2017; Järvi et al. 2018), but differ in their approach to linking knowledge among agents: knowledge ecosystems are non-pluralistic whereas knowledge ecologies embrace a pluralistic approach to knowledge. Consensus building across different disciplines may introduce considerable inertia to the research process (Duncan et al. 2020; Pohl and Hirsch Hadorn 2008). Defining knowledge ecologies as a set of discipline-specific knowledge ecosystems overcomes this inertia by allowing researchers to contribute their knowledge without needing to engage with other disciplines themselves (as illustrated by Case Study 1).

The modular knowledge ecology presented in Fig. 1 may be viewed as a collection of discipline-specific knowledge ecosystems that combine to achieve collaborative actions (high-level research or management outcomes requiring knowledge from multiple disciplines) (ref?). Knowledge ecosystems permit efficient knowledge generation and curation within disciplines (Järvi et al. 2018). The logical space between disciplines is addressed within collaborative actions and the integration of multiple knowledge ecosystems withing a single knowledge structure ensures plurality. Thus, the modular knowledge structure presented in Fig. 1 satisfies the three axioms of transdisciplinarity outlined by Nicolescu (2010).

Anatomy of a knowledge ecosystem

Knowledge ecosystems require a common knowledge base (Järvi et al. 2018) to provide a common language for information sharing between researchers and effective communication between researchers and non-researchers (Fig. 2), a point which has recently been emphasised for invasive species (Wallace et al. 2020). This streamlines assessment of research relevance for non-researcher priorities (Dietze et al. 2018), empowering non-researchers to identify conceptual or data deficiencies or improve the relevance of outputs (e.g. forecasts, scenario analyses) to their priorities (Antunes et al. 2006).

To fully realise the benefits of a shred knowledge base, all agents need to understand its core concepts and how these are implemented using models and available input data. This vision for knowledge ecosystems corresponds closely to the extended peer community approach to transdisciplinary research in the typology of Popa et al. (2015) by "integrating scientific and extra-scientific expertise from the relevant stakeholder communities". The extended peer community approach is an attempt to implement post-normal science principles, which emphasise the benefits of knowledge held by non-scientists to the science process itself (Funtowicz and Ravetz 1993). In practice, the long-term functioning of such a knowledge ecosystem might involve periodic mediated modelling exercises where "stakeholders…collaborate together in the development of a simulation model about a specific problem" (Antunes et al. 2006).

Building a spread and impacts knowledge base

In this section and the two case studies, we focus on the *spread and impacts* knowledge ecosystem (Fig. 2 and bottom left of Fig. 1), since this covers subject matter that most biological invasions researchers are familiar with. We begin by outlining the knowledge base supporting forecasts. We then illustrate how developing a shared knowledge base facilitates the rapid updating of forecasts in light of new research and stakeholder feedback.

What the knowledge base should include

Wilding conifer invasions may be viewed as complex systems, with multiple interacting processes—fecundity, dispersal, vulnerability of receiving environments, and atmospheric fluid dynamics (Buckley et al. 2005; Caplat et al. 2012)—potentially leading to chaotic dynamics in spread and infill rates (Muthukrishnan et al. 2018). The potential for chaos means that a statistical approach to modelling spread and infill rates (i.e. fitting spread and infill coefficients to data obtained from a small set of case studies) is unlikely to accurately predict invasion patterns (Pyšek and Hulme 2005). Rather, it is more informative to treat conifer spread and infill rates as emergent properties of mechanistic models incorporating interactions between the main processes driving invasion (Lewis 2016; Muthukrishnan et al. 2018). The challenge with this approach is that accurate data for these processes, and interactions between them, are difficult to obtain. Consequently, there may be considerable uncertainty about parameters regulating key processes in these models, and a range of simplifying assumptions will be required, especially during the initial model

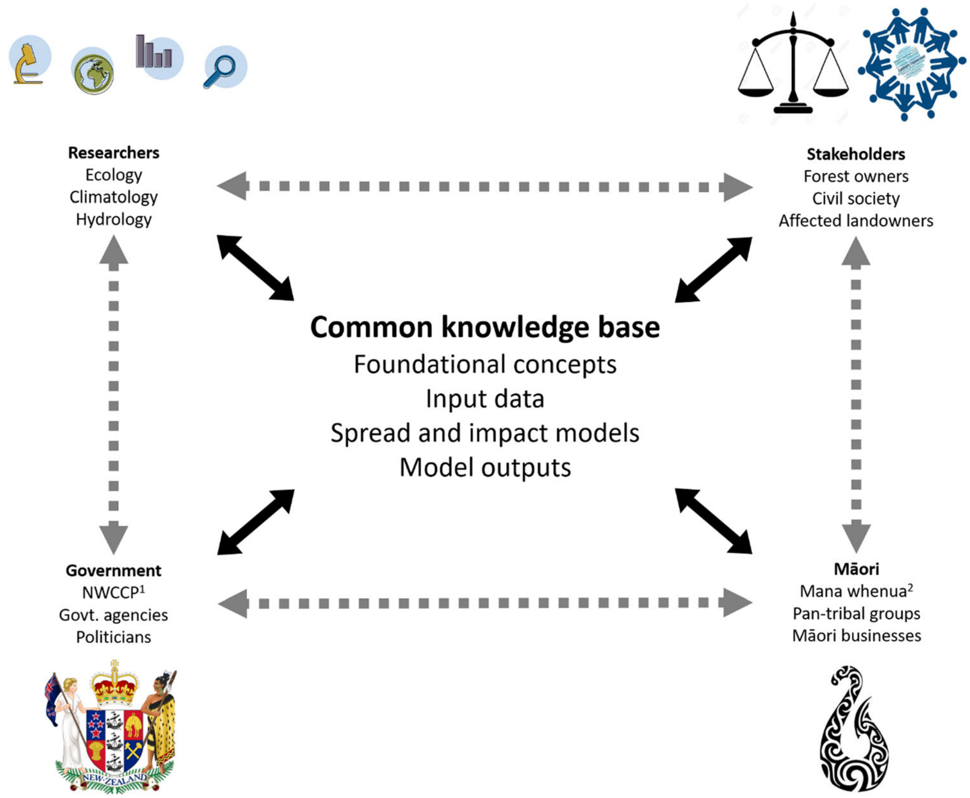

Fig. 2 Overview of a knowledge ecosystem for wilding conifer spread and impacts in New Zealand. The black solid arrows link each agent group to a common knowledge base. All groups both access and modify the knowledge base and all groups adapt their behaviour in accordance with the common knowledge base. The dashed grey arrows acknowledge potential or existing links between all agent groups, independent of the shared knowledge base. [1]National wilding conifer control programme: https://www.mpi.govt.nz/protection-and-response/long-term-pest-management/wilding-conifers/. [2]Māori groups with kaitiakitanga (guardianship) responsibilities for an area of land or a water body (Makey and Awatere 2018)

construction phase. However, the great advantage of this mechanistic approach is that new data on key processes produced by individual research projects can be rapidly incorporated into the knowledge base (as illustrated in Case Study 1 below). More generally, there is a growing realisation of the importance of constructing models to permit iterative (updated) forecasts as new data are gathered (Dietze et al. 2018), and we view a mechanistic basis for forecasting models as vital to achieving this.

A summary of the spread and impacts knowledge base

Below we outline the common knowledge base in the *spread and impacts* knowledge ecosystem, spanning scientific concepts, literature, data sources, and modelling techniques. A full text description as well as all data and scripts used are provided in the supplementary material.

We used *Pinus nigra* as our model species, since rigorous data are available from Buckley et al. (2005) on seedling establishment, juvenile survival, and fecundity. Buckley et al. (2005) provide different seedling and juvenile survival rates for grassland and shrubland, survival rates in shrubland being lower due to shade intolerance of *P. nigra* seedlings, while also allowing for negative impacts of herbivory. Buckley et al. (2005) identified two classes of adult with contrasting fecundity rates. We assigned adults to one of the two fecundity rate classes identified by Buckley et al. (2005), based on conspecific competition and the transition probability data provided by them. Seed production of *P. nigra* in New Zealand follows a mast-seeding pattern, with generally low rates of fecundity punctuated by years of extremely high seed production (Coutts et al. 2012). We incorporated mast seeding by simulating interannual variation in seed production (see supplementary file "MastSeedingSimulation.pdf"). Seed dispersal within 1 ha "sites" followed Buckley et al. (2005), while dispersal within and between 1 km^2 "landscapes" employed the WALD model (Caplatt et al. 2012), to incorporate the effects of variation in wind speed and seed traits on dispersal.

We used infestation data from the Land Information New Zealand wilding conifer infestation database (WCIS), which reports infestation data for 1 km × 1 km grid squares (referred to here as "landscapes") as four different density classes: outlier (< 8 trees ha^{-1}), sparse (8–150 trees ha^{-1}), intermediate (150–600 trees ha^{-1}) and dense (> 600 trees ha^{-1}). This data format has the disadvantage that basic invasive processes—establishment, competition, dispersal, and stand development—occur at much smaller scales than 1 km^2 (e.g. Caplat et al. 2014). To deal with this, we adopted a multi-scale approach to modelling wilding conifer invasion. This involves modelling (a) infilling within invaded 1-ha "sites" (Fig. 3), (b) dispersal between sites within 1-km^2 landscapes (Fig. 4), and (c) dispersal from invaded landscapes to neighbouring landscapes (Fig. 5).

We use the Landcover Database (LCDB) version 4[3] to describe landcover at a resolution of 100 m. We reclassified LCDB classes into "grass", "shrub", and "other" categories. We assumed negligible conifer invasion occurs in the "other" category. Landcover information was incorporated in estimating landscape-level infill rates (via a series of matrix multiplications) and invasion of downwind landscapes (by intersecting downwind seed rain density in 100 m distance bands with land cover categories). Wind data were obtained from Leathwick et al. (2003).

Fixed control costs were assigned to each density class on a cost-per-hectare basis: outlier, \$2 ha^{-1}; sparse, \$10 ha^{-1}; intermediate, \$1200 ha^{-1}; dense, \$2500 ha^{-1} (Jonothan Underwood, pers. comm.) These costs assume helicopter-based spot-spraying for outliers, ground-based control for sparse and intermediate invasions, and aerial boom spraying for dense sands. We split control costs into removal (i.e. control of conifers in invaded 1-km^2 "landscapes") and containment (i.e. periodic control of downwind invasion into neighbouring 1-km^2 landscapes).

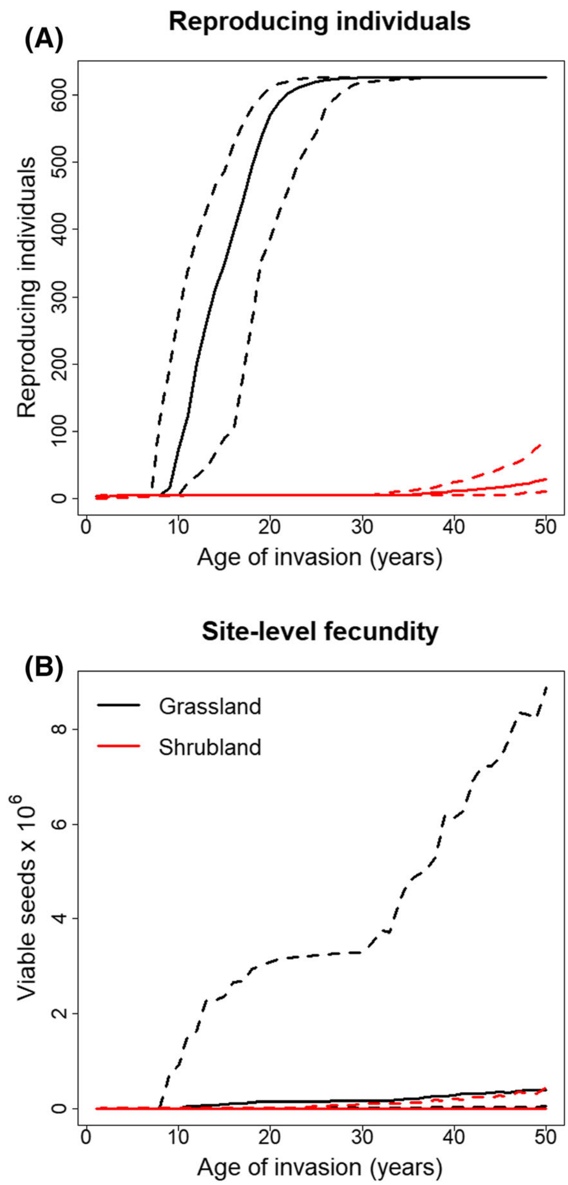

Fig. 3 Median (solid lines) and 95% confidence limits (dashed lines) for the number of reproducing individuals **a** and total seed output **b** for 1 ha sites receiving low levels of external colonisation (on average 5 individuals per year). These results assume a mean of seven years till first reproduction. We used the upper bound (97.5th percentile) fecundity values for invasion ages 10, 20 and 50 in grassland in the downwind colonisation examples presented in Fig. 4b, c

[3] https://lris.scinfo.org.nz/layer/48423-lcdb-v41-land-cover-database-version-41-mainland-new-zealand/.

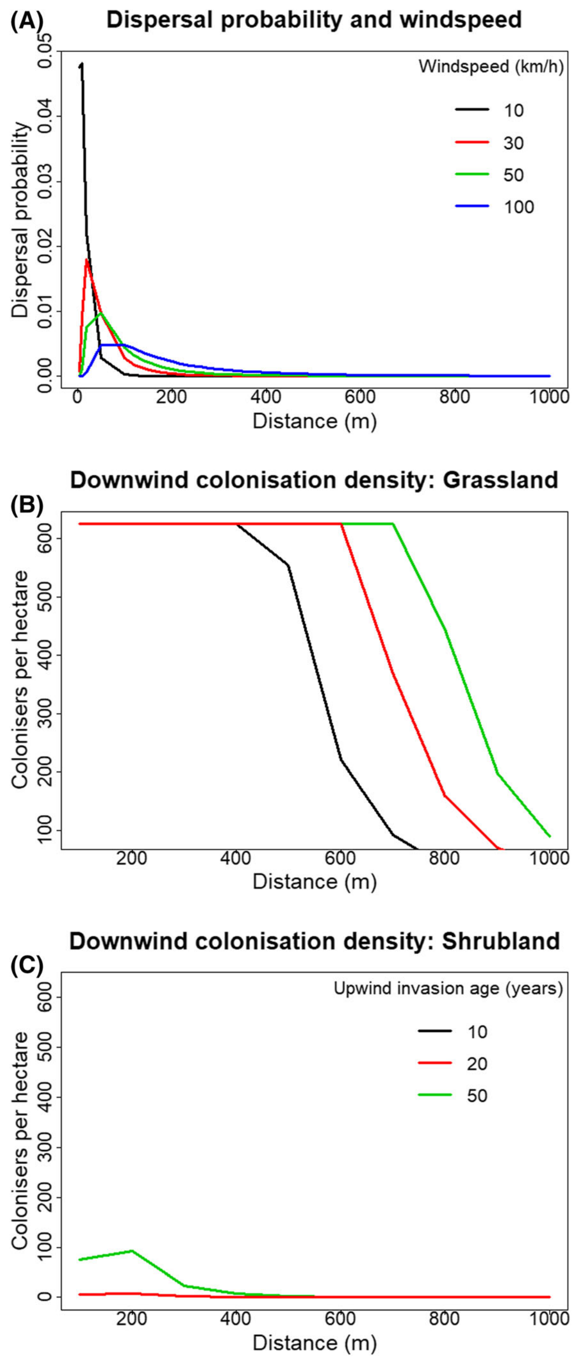

Fig. 4 Effect of wind speed on dispersal probabilities using the WALD equation as parameterised in the between-site and between-landscape spread models (**a**); and colonisation densities in grasslands (**b**) and shrublands (**c**) under "worst-case" (i.e. 97.5th percentile) seed rain values from upwind 1 ha sites with invasions aged 10, 20 and 50 years, assuming mean wind speed of 30 km hour^{-1}. Maximum densities were set at 625 trees per hectare, which is the number of colonisation sites available in the infill model

We estimated return on spend on removing existing invasions as a proportion of removal cost to the sum of future removal and containment costs:

$$\text{Return on spend} = \frac{\text{Removal}_{t2} + \text{Containment} - \text{Removal}_{t1}}{\text{Removal}_{t1}}$$

where: *Removal$_{t1}$* is the initial removal cost, *Removal$_{t2}$* in the potential removal cost at the end of the time period, and *Containment* is the cost of containing downwind invasion of neighbouring landscapes. A return on Spend of 1 indicates that removal of existing invasions results in averted additional control costs equivalent to the initial removal cost. In our scenario analyses, we calculate averted control costs and ecosystem impacts from removal of initial invasions for a 21-year time period (equivalent to three *Pinus contorta* "generations"). We also calculate the annualised percent rate of return on investment (for a 21-year investment period) using the standard compound interest-based method:

$$r = 100\left[\left(\frac{\text{Removal}_{t2} + \text{Containment} + \text{Removal}_{t1}}{\text{Removal}_{t1}}\right)^{\frac{1}{21}} - 1\right]$$

In this study we assume that all invasions from currently invaded 1-km^2 landscapes to neighbouring downwind landscapes are contained. Thus, potential ecosystem impacts in the absence of removal are estimated as a function of potential infilling in invaded landscapes. We modelled maximum potential wilding conifer impacts on biodiversity, water yield, and erosion reduction following Mason et al. (2016). Directly proportional relationships between wilding conifer density and impacts on water yield and erosion reduction were assumed (i.e. the product of observed density expressed as a proportion of maximum density and wilding conifer impacts at maximal density—625 trees per hectare). Maximum potential impact of wilding conifers on livestock-carrying capacity was based on carrying capacity data provided by the Land Resource Inventory of New Zealand, with carrying capacity of zero assumed at maximum conifer density (Newsome 1992). Density-impact relationships for biodiversity and stock units were modelled using a power function of conifer density (expressed as a proportion of maximum density), to reflect the fact that conifer impacts increase rapidly at low densities for these ecosystem services (e.g. Dickie et al. 2011).

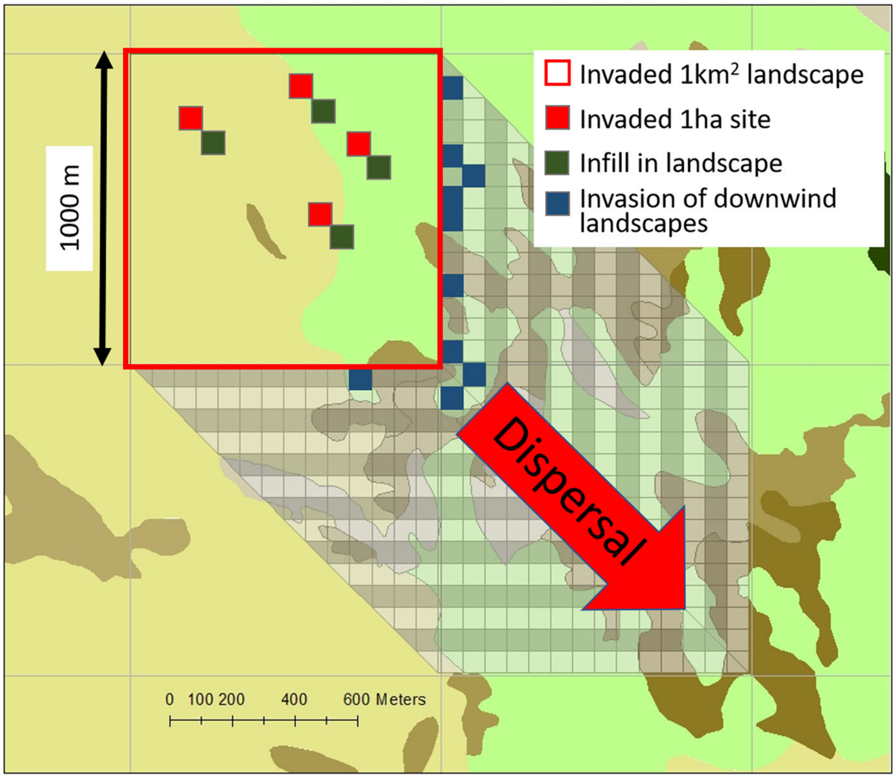

Fig. 5 Visual representation of multi-scale invasion model. Conifer invasion data are available for invaded 1-km^2 grid squares (termed "landscapes" for brevity; large open red square). We model **a** infill (increases in conifer stand density) within invaded 1 hectare "sites" (small solid red squares), **b** infill within invaded landscapes (i.e. invasion of non-invaded sites from invaded sites; small solid green squares), and **c** downwind invasion of neighbouring landscapes (small solid blue squares). Background colours represent different land cover types, which were used as an indicator of vulnerability to invasion. Overlaid light and dark shading indicate 100-m dispersal distance bands. These bands are intersected with landcover data to estimate downwind colonisation density

We used "head fire intensity", which is a measure of energy output per metre length of the most rapidly advancing fire front (the head fire), as our metric of increased wildfire impact, since this is the aspect of fire behaviour most likely to be affected by the increased fuel load provided by wilding conifers. We used models from Clifford et al. (2013) and Briggs et al. (2005) to make spatial predictions of changes in head fire intensity caused by each wilding conifer density class.

Spread and impacts forecasts and scenario analyses

In this section we describe a selection of outputs derived from the common knowledge base as presented to stakeholders. This includes a) cost–benefit analyses focussed on the averted future control costs arising from removal of infestations in the near future, and b) a rudimentary multi-criteria assessment aimed at minimising trade-offs between financial cost–benefit and ecosystem impacts in prioritising wilding conifer removal effort.

Prioritising allocation of funding to 1-km^2 landscapes with the highest return on spend was a more efficient strategy for reducing containment costs (i.e. produced more rapid increase in averted containment costs with cumulative initial removal costs) than prioritising funding to landscapes with the largest containment costs (Fig. 6a). Indeed, with this strategy, 70% of total containment costs could be averted for a tiny fraction of the cost of removing all existing infestations (see intersection point between red curve and dotted horizontal line at 70% of total containment costs in Fig. 6a). Similar results were obtained for

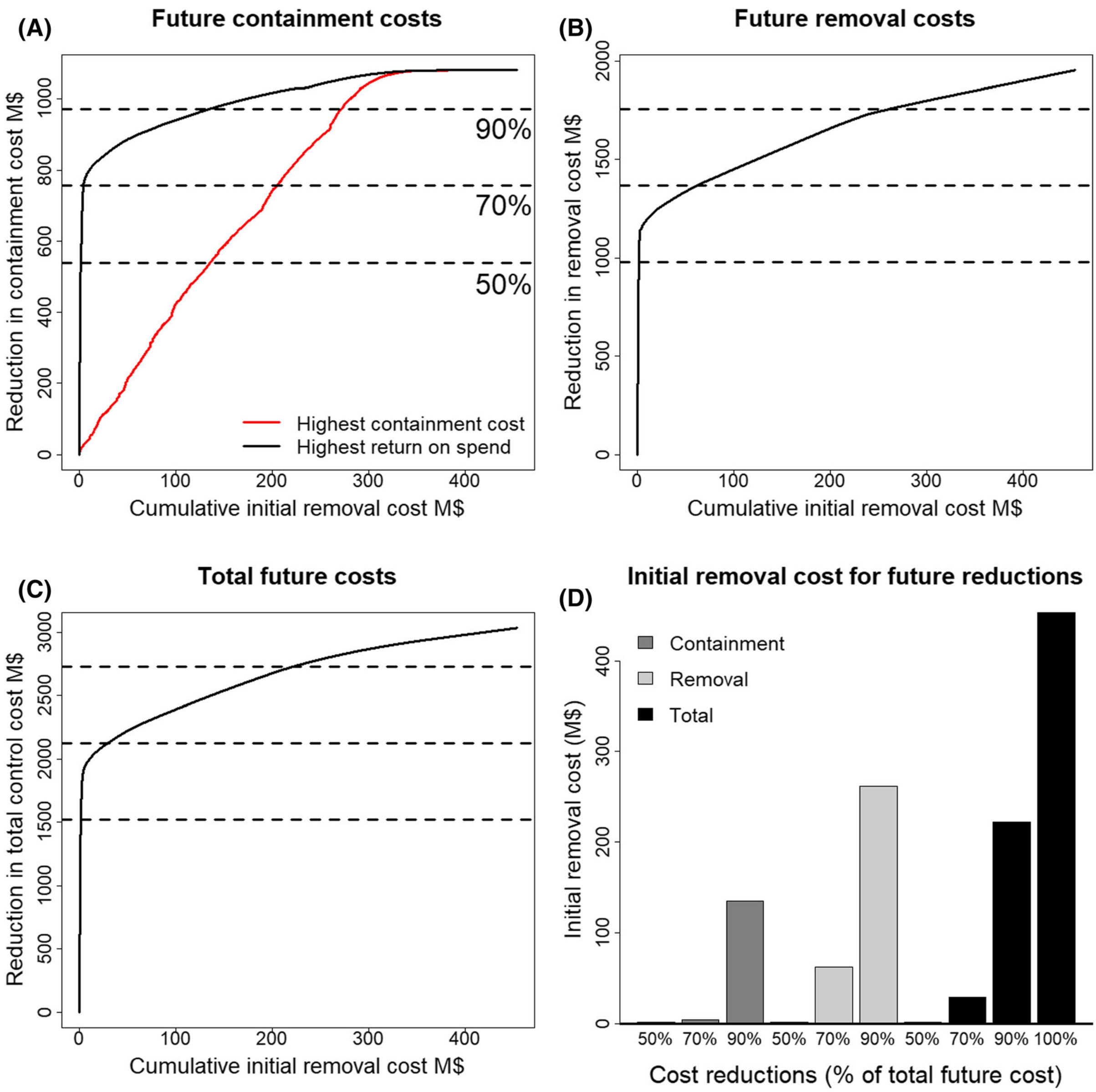

Fig. 6 Estimated initial removal costs required to reduce future containment (**a**), removal (**b**), and total control costs (**c**). The solid red and black curves represent, respectively, scenarios where control effort is prioritised to areas where containment costs are highest or where return on spend is highest. Horizontal dashed lines indicate 50, 70, and 90% reductions in future control costs. Removal costs required to achieve 50, 70, 90, and 100% reduction in control costs under a strategy prioritising 1-km^2 landscapes with highest return on spend are presented in the bar graph (**d**)

future removal costs and total control costs following a strategy prioritising return on spend (Fig. 6b–d).

Annualised rates of return on initial removal spend for 50% and 70% total reduction in control costs (respectively 39.2% and 22.7% *per annum*) were much higher than could be expected based on the current cost of money (Fig. 7). Even the return rate for 90% and 100% reduction in future control costs (respectively 13.1% and 10.2% *per annum*) compare favourably with the New Zealand treasury range of discount rates (from ~ 2% for zero-risk investments to 9% for normal market risk levels). The low risk end of this range is probably the most relevant comparison for investment in environmental management since

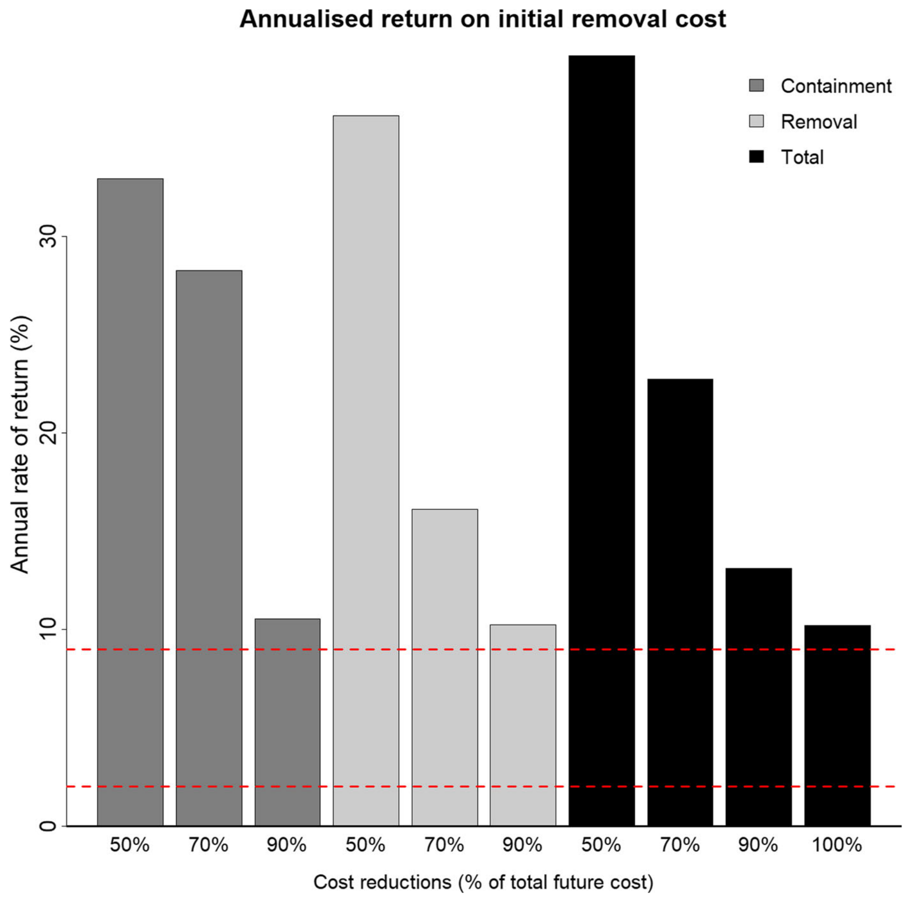

Fig. 7 Annualised rate of return in reduced control costs for initial removal costs required to achieve 50, 70, 90 and 100% reduction in total control costs. Results assume a scenario where management effort is prioritised to maximise return on spend. Red dashed lines indicate New Zealand Treasury Guidance on discount rates (expected real returns) for risk free (i.e. government bonds—lower line) and high risk (i.e. share market—upper line) investments

"returns" are very unlikely to be correlated with fluctuations in markets.

Prioritising management to achieve multiple objectives

It is extremely difficult to devise management strategies to optimise outcomes for any single criterion when multiple criteria must be considered. Sub-optimal outcomes for any single criterion in multi-criteria assessments are often termed trade-offs (Mason et al. 2016). Trade-off minimisation analyses are a "valueless" (beyond the obvious value judgement of which criteria should be included) approach to multi-criteria assessment, where no a priori weights are applied to competing criteria. We applied the method of Mason et al. (2016), which fits weights to competing criteria that minimise trade-offs between priorities (Fig. 8). Our results show that allocating heavier weights to avoiding biodiversity loss and, to a lesser extent, increases in erosion reduces overall trade-offs. The high weightings applied to biodiversity and erosion are to be expected since priorities where benefits have a highly right-skewed probability distribution (i.e. many low and few very large values) are most prone to trade-offs (Mason et al. 2012). Incidentally, this aligns with recent survey-based work indicating that by far the highest priority for people in New Zealand is the protection of indigenous-dominated ecosystems (Turner 2019).

Case Study 1: Researcher influence on knowledge base

In this section we provide an example illustrating how a knowledge ecology composed of discipline specific knowledge ecosystems empowers researchers to contribute to long-term management objectives and strategies, with a specific focus on improvements to the knowledge base of the *spread and impacts* knowledge ecosystem (Fig. 2). We do this by

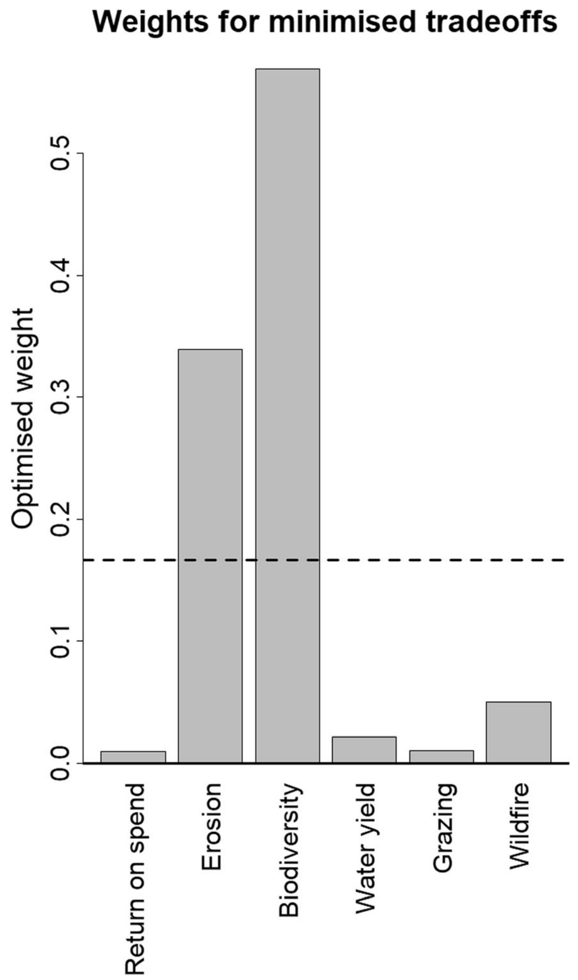

Fig. 8 Fitted weights (i.e. contribution to the weighted mean benefit of removing wilding conifers from invaded 1-km^2 "landscapes") applied to competing criteria to minimise tradeoffs between priorities. The horizontal dotted line indicates the weight value if all criteria were allocated the same weight (i.e. all criteria make an equal contribution to mean benefit calculations)

demonstrating how new data on seed dispersal, generated from a single research study, could lead to rapid updating of spread (and hence impact) predictions, via an open-access, transparent invasive spread model.

Intraspecific variation in seed dispersal

Seed dispersal is a key component of the wilding conifer invasion process. Seed dispersal in family Pinaceae is generally wind-assisted, with winged seeds (known as samara Wyse et al. 2019) falling with a spiralling motion. The rate of descent in still air once the samara enters this spiralling motion is known as terminal velocity (Wyse et al. 2019). Terminal velocity and windspeed interact to influence dispersal distance. In the simplest scenario (assuming no turbulence or variation in wind speed with height above the ground), samara dispersal distance is a function of horizontal wind speed, seed release height and the inverse of terminal velocity (Wyse et al. 2019). Put simply, the harder the wind blows, the greater the height the samara falls from and the slower it falls, the further the samara will travel upon release from the cone.

Terminal velocity decreases as the ratio of wing area to samara mass increases (Wyse et al. 2019). This ratio, and consequently terminal velocity, can vary markedly within populations, individual trees and even individual cones. Wyse et al. (2019) developed a protocol for measuring samara terminal velocity under controlled conditions and with a high level of repeatability. They used this protocol to explore variation in terminal velocity in *Pinus radiata* populations in New Zealand.

We combined the terminal velocity data collected by Wyse et al. (2019) with the WALD dispersal probability equation (Caplat et al. 2012) to quantify the effect of incorporating intraspecific variation in terminal velocity on dispersal probability estimates (like those displayed in Fig. 4a). Here, we compare dispersal probabilities generated using (a) a dispersal-distance curve calculated using mean terminal velocity value for *P. radiata* or (b) by aggregating dispersal-distance curves calculated separately for each individual terminal velocity value recorded by Wyse et al. (2019). These exploratory analyses show that under certain conditions—high wind speeds and high seed release heights—using the mean terminal velocity may greatly underestimate distances for high ($\geq$ 95th) dispersal percentiles (i.e. underestimate the length of the dispersal probability tail. In more concrete terms, using the mean value underestimates the distance within which 95% of seed disperses. Figure 9 documents the difference in distance estimates for each quantile (i.e. distance containing x % of the total dispersal probability), when summing dispersal probability curves across individual terminal velocity records versus a probability curve generated using the mean terminal velocity value.

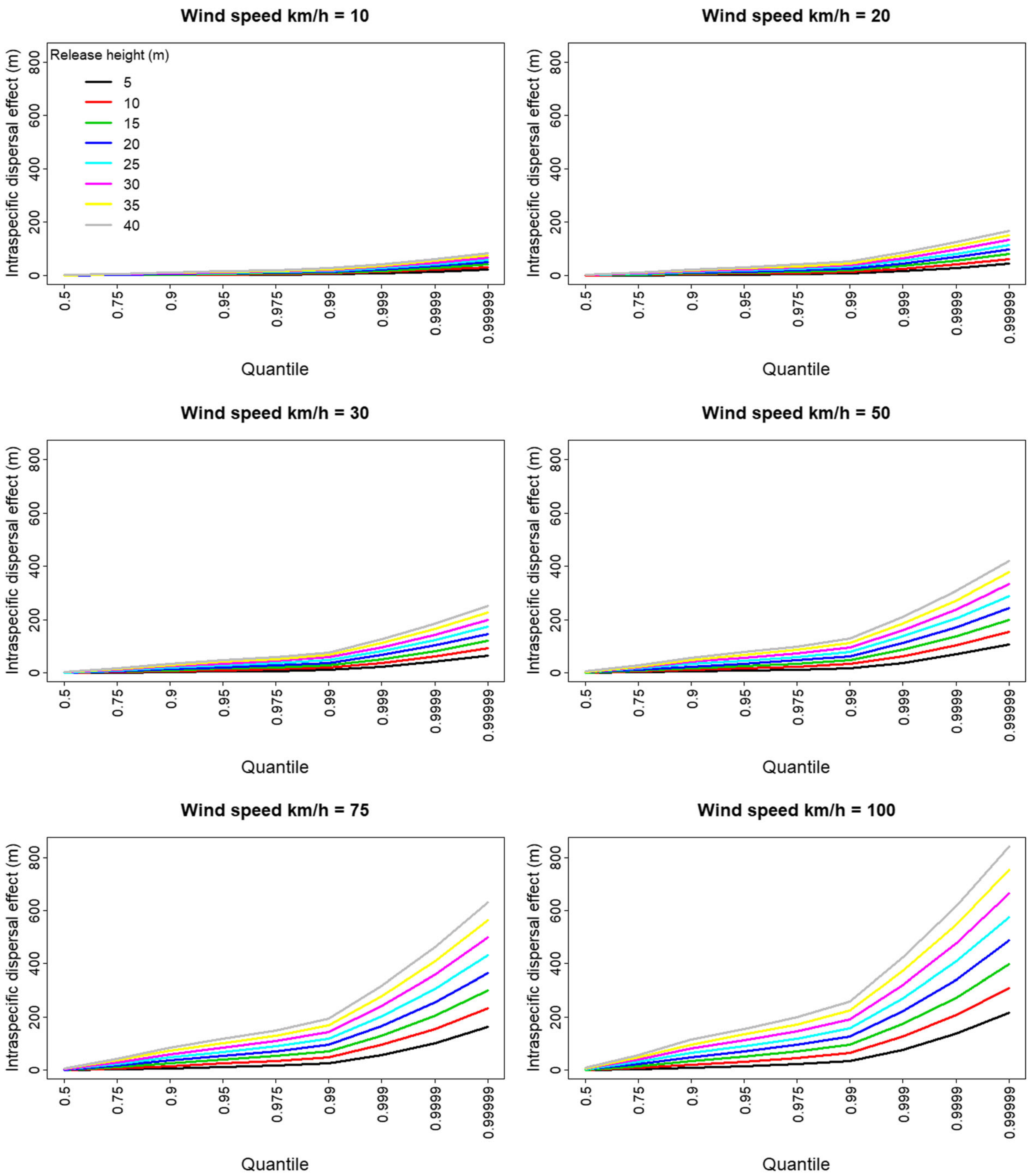

Fig. 9 Effect of incorporating intraspecific variation in seed terminal velocity on dispersal probability estimates for *Pinus radiata* under various combinations of horizontal wind speed and seed release height. The vertical axes document the difference in distance estimates for each quantile (i.e. distance containing x % of the total dispersal probability), when summing dispersal probability curves across individual terminal velocity records *versus* a probability curve generated using the mean terminal velocity value. Terminal velocity data were obtained from Wyse et al. (2019)

Using the mean terminal velocity of a species could have two major consequences for modelling invasion processes: first, the area at risk of invasion from existing *P. radiata* stands is likely to be underestimated; second, the flatter, longer dispersal curve produced when incorporating intraspecific variation might lead to more rapid population increases from downwind dispersal, because a smaller proportion of seed will be subject to "futile" dispersal (i.e. dispersal to areas where colonisation rates exceed carrying capacity). However, it is difficult to quantify these effects without information on stand-level fecundity and seedling establishment rates. Employing the downwind invasion components of the Spread and Impacts knowledge ecosystem provides a convenient way to quantify these effects. Figure 10 documents the effect of incorporating intraspecific variation in seed terminal velocity on estimates of potential colonised area, while Fig. 11 illustrates the effect of incorporating intraspecific variation in seed terminal velocity on estimates of effective population increase.

Reflections on case study 1

This case-study demonstrates how and open-access invasive spread modelling platform can provide researchers with evidence to justify the updating of spread and impact forecasts in accordance with their research findings. Specifically, our results show that not including intraspecific variation in terminal velocity in forecasting might lead to severe underestimation of future wilding conifer spread. This has two main implications for the knowledge base of the Spread and Impacts knowledge ecosystem. First, it suggests we should document both within and between-species variation in terminal velocity for the main wilding conifer species. Second, it shows that forecasting models must be able to accommodate intraspecific variation in terminal velocity in estimating down-wind invasion from existing wilding conifer infestations.

Case Study 2: Non-researcher influence on knowledge base

This section documents a collaborative process we employed to include non-researcher perspectives in structuring the *Spread and Impacts* knowledge base (Fig. 2). This exercise used a practical goal—production of forecasts to support wilding conifer management in a specific management unit (the upper Waihopai and Wye catchments, Fig. 12)—as a vehicle for integrating non-researcher perspectives into the knowledge base. The non-researcher group was the Southern Marlborough Regional Wilding Conifer Steering Group, which is a regional governance body within the National Wilding Conifer Control Programme, tasked with prioritising resources for wilding conifer control within their region. This group includes affected landowners, representatives from central and local government (the Department of Conservation, Ministry for Primary industries and Marlborough District Council), members of a local non-governmental group (the South Marlborough Landscape Restoration Trust), and a representative of a national conservation-focussed non-governmental organisation (The Royal Forest and Bird Society of New Zealand). Notably lacking from the group were any Māori perspectives.

We applied an iterative focus groups approach (Nyumba et al. 2018) to incorporating non-researcher perspectives. This involved repeatedly exposing the modelling approach and model outputs to members of the stakeholder group, recording feedback and updating models and model outputs in accordance with this feedback. Stakeholder feedback was recorded during discussions and draft summaries of these records were shared with members of the stakeholder group to ensure that they represented a common understanding of the topics discussed. In the summary of this process presented below, we focus on instances when stakeholder feedback resulted in changes to the modelling approach and model outputs, or highlighted urgent priorities for future research.

Project planning, communicating results and recording stakeholder feedback

The process began with a field reconnaissance trip (to assess the accuracy of infestation data in the Waihopai catchment and obtain landowner knowledge on the local history of conifer invasions) and resulting discussions to identify relevant forecasting outputs. After this initial contact, a meeting was organised to share forecasting results with relevant stakeholders and record their feedback.

Ultimately, two meetings were convened (the second including a larger number of stakeholders). The lead author of our study explained the spread

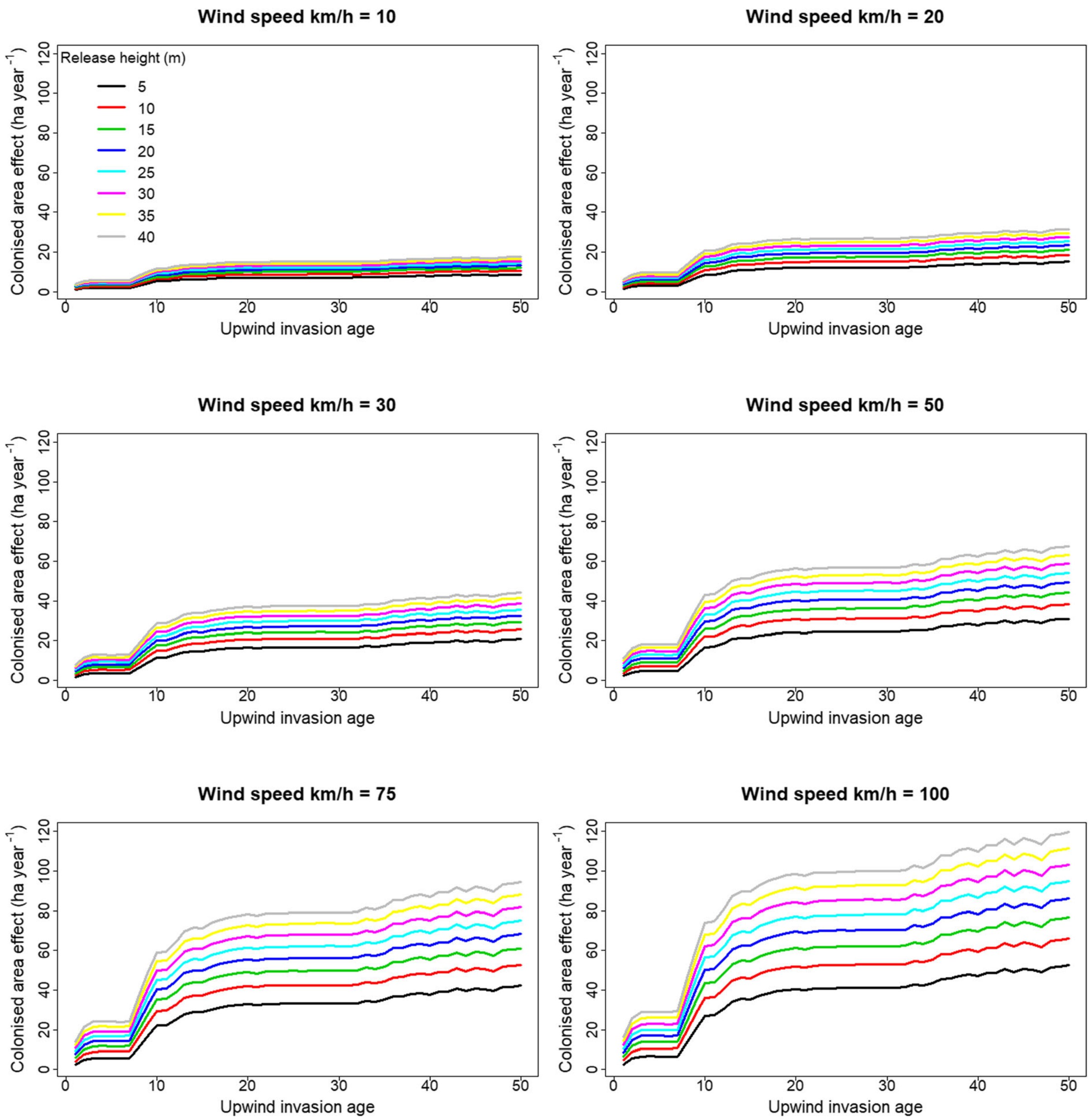

Fig. 10 Effect of incorporating intraspecific variation in seed terminal velocity on estimates of potential colonised area (i.e. area receiving at least 1 successfully establishing individual per hectare, per year) for *Pinus radiata* under various combinations of horizontal wind speed, seed release height and upwind invasion age. The basic scenario assumes a maximum dispersal distance of 10,000 m and an invasion front (perpendicular to the prevailing wind) of 1000 m. Vertical axis values indicate the excess estimated area obtained when incorporating intraspecific variation relative to using the mean value

modelling approach (including an animated version of Fig. 5) and presented forecasting results (including those in Fig. 12). These results were used to identify the areas within the Waihopai-Wye management unit with the highest return on spend for removal of existing invasions. Maps of wind speed and vulnerability of habitat to invasion (Fig. 12d and e) were presented to aid interpretation of results.

These results were used to propose a broad strategy, focussing removal effort on low-density infestations where the return on removal spend is highest. In particular, return on spend was compared for

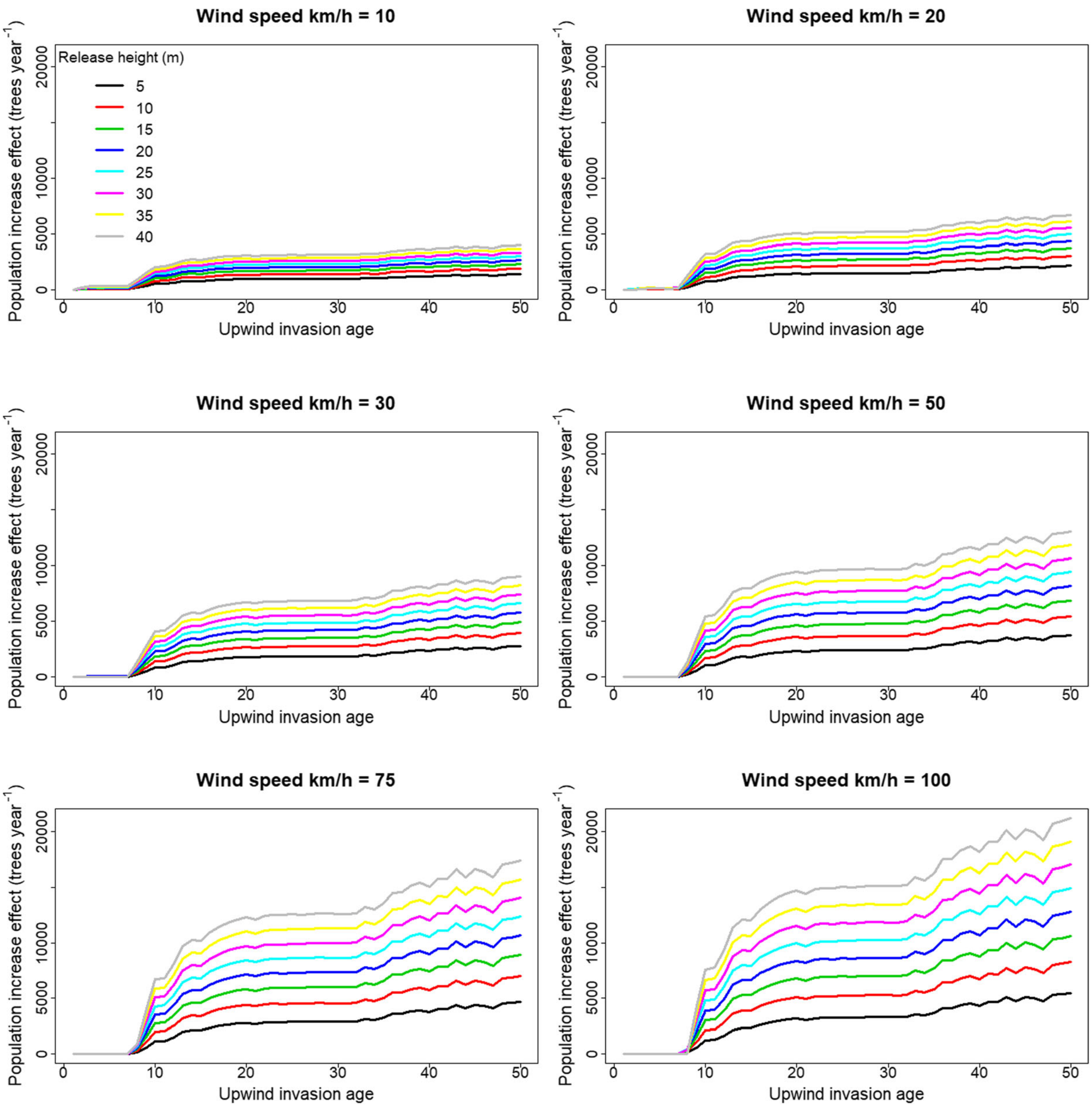

Fig. 11 Effect of incorporating intraspecific variation in seed terminal velocity on estimates of effective population increase for *Pinus radiata* under various combinations of horizontal wind speed, seed release height, and upwind invasion age. The basic scenario assumes a maximum dispersal distance of 10,000 m and an invasion front (perpendicular to the prevailing wind) of 1000 m

individual operation zones in the Waihopai-Wye management unit (Table S1), and between management units across the wider region (southern Marlborough) and neighbouring areas (Table S2).

Stakeholders identified two main conceptual deficiencies in knowledge base. First, the modelling approach made no attempt to document the potential impact (both in terms of removal costs and ecosystem impacts) of low-probability, long-distance colonisation events that are known to occur in the region. Second, the assumption that seed dispersal always occurs in the direction of the prevailing wind (from north-west to south-east) was flawed. In response to this feedback, we committed to (a) producing forecasts

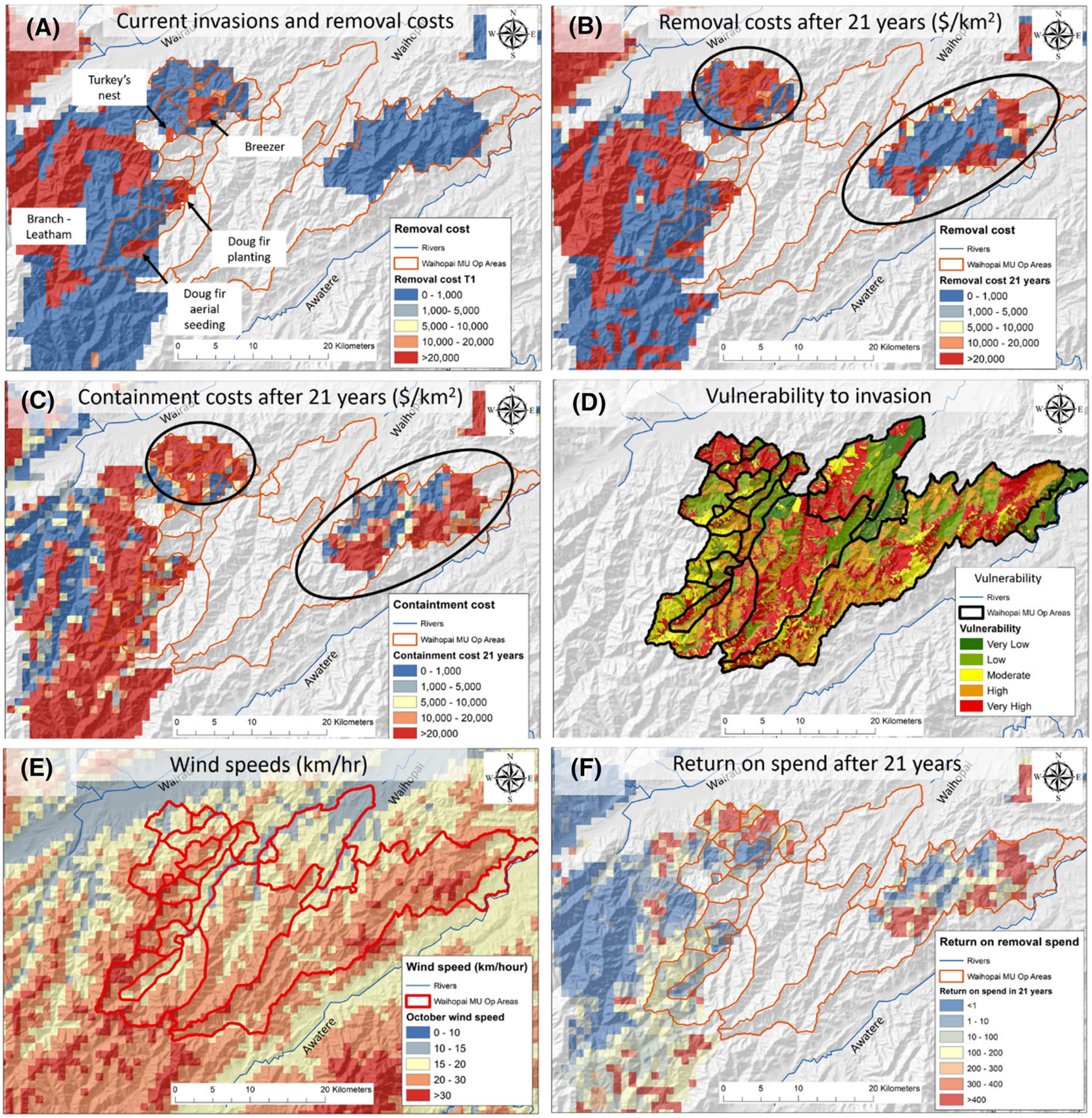

Fig. 12 Initial (**a**) and future (**b**) removal costs, containment costs (**c**) and return on removal spend (**f**) for currently invaded 1 km^2 "landscapes" in the Waihopai and Branch-Leatham management units. Areas indicated by text boxes in (**a**) are invasion sources stemming from historic planting and seeding trials. "Doug fir" refers to Douglas fir, *Pseudotsuga menziesii*. Black ellipses indicate areas where large increases in removal costs and high containment costs (due to vulnerable habitats—**d** and high windspeeds—**e**) drive high return on spend

of removal costs and ecosystem impacts arising from low-density (0.25 colonisers ha^{-1}) colonisation events; and (b) updating the downwind dispersal model to incorporate variation in the frequency and speed of wind blowing from all cardinal and intercardinal directions.

Another major conceptual concern was the feasibility of maintaining effective containment operations if significant seed sources were allowed to persist in the region. Most stakeholders knew that dispersal events spanning tens of kilometres can occur under extreme conditions, meaning that effective containment would require a region-wide surveillance strategy. From the ensuing discussion it emerged that urgent research was needed on the cost, reliability and integrated deployment of multiple detection technologies before analyses of realistic management scenarios would be possible. This is a complex research problem requiring input from the *management technologies* (i.e. cost and reliability of detection technologies) and *operational logistics* (i.e. integrated deployment of multiple detection technologies) knowledge ecosystems as well as the *spread and impacts* knowledge ecosystem (Fig. 1). This outcome of the collaborative process reveals the need for clear integration pathways between knowledge sources via collaborative actions (Fig. 1), since stakeholders were effectively driving a transdisciplinary research agenda by demanding input from three different knowledge ecosystems to inform a long-term management strategy (Fig. 1).

Two areas of data deficiency were also noted. First, infestation data for one of the main management units of Southern Marlborough (Awatere) were not included in the infestation database. Stakeholders noted that this was due to individual landowners withholding data access for fear of incurring liabilities related to wilding conifer management. Second, the most recent spatial data on threatened species and ecosystem distributions were not included in analyses of biodiversity impacts. This is especially important in Marlborough since it is a major centre of endemism in New Zealand, and many threatened species occur in ecosystems that are highly vulnerable to conifer invasion (Jan Clayton-Greene pers. comm.). The relevant stakeholders committed to addressing these data deficiencies to improve the accuracy of forecasts.

Despite the conceptual and data deficiencies in the knowledge base they identified, stakeholders believed the forecasting approach we used could be a useful source of information in devising long-term management objectives and scenarios for their region. Indeed, stakeholders subsequently used the results in Fig. 12f to identity management priorities in obtaining funding from the National Wilding Conifer Control Programme (Tessa Roberts. pers. comm.).

Reflections on case study 2

This collaborative process represents an expanded peer community approach to transdisciplinary research (Popa et al. 2015). Our process conforms to the "extending the scientific peer community and for integrating multiple legitimate perspectives into the scientific analysis" criterion of extended peer communities (Popa et al. 2015). It also corresponds very closely to the mediated modelling approach (Antunes et al. 2006) to participatory decision making, since stakeholders were able to demand fundamental changes to the modelling process.

Based on this experience, an expanded peer community approach appears well suited to encouraging participation of non-researchers in enhancing the knowledge base at the centre of each knowledge ecosystem. The feedback we received from stakeholders reveals several key insights. Firstly, non-researchers thoroughly comprehended the scientific information we presented. Second, non-researchers were highly motivated to engage with technical information because they could see the relevance of that information to their own objectives. Finally, non-researchers displayed a high level of confidence in suggesting improvements to the modelling process and identifying future research priorities. This last point is crucial since it shows that non-researchers have a vital role in the development of the knowledge base by (a) using their lived experience and local knowledge to identify deficiencies in the knowledge base; and (b) providing researchers with guidance on how to prioritise future research. The benefits of an extended peer group approach were recognised at the beginning of the post-normal science movement in the early 1990s: "The extension of the peer community is then not merely an ethical or political act; it can positively enrich the processes of scientific investigation." (Funtowicz and Ravetz 1993).

The lack of any Māori perspectives represents a major gap in our extended peer group. We view this primarily as a consequence of excluding Māori from "Key Participants in Wilding Conifer Management Governance" within the national wilding conifer management strategy (MPI 2014). Māori are increasingly participating and collaborating in resource management decision-making (Harmsworth et al. 2016; Ruru et al. 2017; Te Aho 2010). However, there are still widespread difficulties in meeting the needs and aspirations of Māori, often reflecting factors such as lack of genuine commitment from researchers, mistrust, lack of respect, lack of knowledge, low capacity, and lack of resources (Harmsworth and

Awatere 2013). At least one of the eight iwi claiming kaitiakitanga (authority over environmental decision-making) in the region (Kai Tahu) expresses concern over wilding conifer invasions within their long-term environmental plan,[4] while another (Te Ati Awa) mentions concerns over "weeds" without mentioning wilding conifers specifically.[5] This suggests Māori do have an interest in the outcomes of wilding conifer management in Southern Marlborough, and their views should be represented in the knowledge base and future decisions on management priorities.

Combining science-based and values-based reasoning through governance

Research organisations and researchers are expected to demonstrate clear implementation pathways to justify the funding they receive (Duncan et al. 2020; Perrings et al. 2011). Robust governance structures will form a key component of most implementation pathways, since they offer forums where scientific evidence can contribute to decision-making processes founded on stakeholder values (Sharma-Wallace et al. 2018). Ideally, it will be the role of government or non-governmental administrative agencies supporting invasive species management to facilitate the establishment of governance bodies (Sharma-Wallace et al. 2018). However, researchers may be required to initiate governance processes where connections between stakeholders are weak. Furthermore, researchers have a role in ensuring that governance frameworks are based on clearly articulated stakeholder values, and that governance bodies have ready access to relevant science (Sharma-Wallace et al. 2018). Therefore, it is useful for individual researchers to have at least a basic understanding of good governance principles.

One stakeholder group included in Case Study 2 (Southern Marlborough Landscape Restoration Trust) lists protection of "…the iconic scenic tussock grassland and alpine landscapes of South Marlborough"[6] as their motivation for engaging in wilding conifer management. This is a very region-specific statement of values, and such place-specific stakeholder motivations for wilding conifer management appear to be common throughout New Zealand (Kirk 2019). Connections to specific places (and indigenous species) are often at the heart of stakeholder value frameworks, particularly for indigenous peoples (Lyver et al. 2019), and a recent review suggests that a shared stake in a particular place is a key motivating factor in multi-stakeholder invasive species management efforts (Graham et al. 2019).

The National Wilding Conifer Programme (NWCCP) is potentially structured to empower local stakeholders in decision-making via its Regional Steering Groups. Indeed, through our interaction with the Regional Steering Group established in Southern Marlborough, outputs from the *Spread and Impacts* knowledge ecosystem were combined with values-based reasoning to influence decisions on allocation of conifer control effort (see Case study 2). However, we recognise that Regional Steering Groups were established without an explicit process for identifying stakeholders or considering how their values might be enacted within decision-making processes. Below we suggest some simple approaches for identifying stakeholders and incorporating their values within local governance bodies. We conclude with considerations on how biological invasions researchers and governance bodies might collaborate via an adaptive governance model.

Who should have a role in governance?

In this study we have noted that the treaty partnership between the New Zealand government and Māori mandates the oversight of bi-cultural governance frameworks in environmental decision-making (Fig. 1, centre-top). However, Māori are not included as "Key Participants" in the national wilding conifer management strategy. Case Study 2 reveals the consequences of this for inclusiveness in decision making, with mana whenua (local Māori groups) apparently not represented in the Southern Marlborough Regional Steering Group. Given the legally defined right of mana whenua groups to influence environmental management decisions, this potentially exposes the regional steering group to legal challenges of its management (New Zealand Government 1991). This case study highlights the need for an explicit focus on identifying relevant stakeholder groups. Influence mapping and visualisation (e.g. Bourne

[4] https://ngaitahu.iwi.nz/wp-content/uploads/2013/08/Te-Runanga-o-Kaikoura-Environmental-Management-Plan.pdf.

[5] https://www.teatiawatrust.co.nz/assets/Uploads/Te-Atiawa-Iwi-Environmental-Management-Plan.pdf.

[6] https://www.marlboroughrestoration.org.nz/.

and Walker 2005) is a widely-used tool for identifying stakeholders linked to a given project. Influence mapping is also useful as a guide for stakeholder engagement in a project, with different approaches usually advised for different categories of stakeholder. Various methods exist for categorising and prioritising stakeholders according to power and degree of interest, the most common being some variant of the "power-interest matrix" (De Mascia 2016), which usually assigns stakeholders into four categories (combinations of high and low power and interest). De Mascia (2016) suggests that most attention be given to stakeholders with high interest and high power, followed by those with high interest but less power. Regarding biological invasions, high interest-high power stakeholders might include government agencies responsible for invasive species management, while high interest-low power groups might include community groups or NGOs concerned about environmental or biodiversity impacts of invasive species.

Another approach is the salience model of Mitchell et al. (1997), which uses three dimensions—legitimacy, power and urgency—to categorise stakeholders. Regarding our experience in Case Study 2, the salience model is probably more applicable than the power-interest matrix, because it accommodates stakeholders, such as mana whenua groups, that have moral and legal authority over decisions (legitimacy and power), but low urgency (even despite potentially high levels of interest) due to competing priorities and lack of capacity (Harmsworth and Awatere 2013). Mitchell et al. (1997) emphasise the dynamic nature of urgency, noting that the perceived urgency of an issue for a given stakeholder group might vary markedly through time. For instance, the perceived urgency of wilding conifers to mana whenua may increase either as competing priorities become less pressing, or they receive information detailing the potential impacts of wilding conifers on their values.

Enacting stakeholder values within governance bodies

The next consideration is understanding how stakeholder values might be enacted in governance frameworks. We are aware of three regions where stakeholder views on wilding conifer impacts have been assessed (primarily through surveys, interviews and participatory mapping, Kirk 2019). However, we are not aware of any explicit efforts to incorporate place-specific stakeholder values in the decision-making processes of Regional Steering Groups.

Collaborative approaches to environmental governance gained prominence in the second half the of the twentieth century, and have been the focus of considerable research activity for decades (e.g. Bulkeley and Mol 2003). Although increased participation of non-state actors in environmental governance is generally viewed as positive for environmental outcomes, recent research has questioned whether this is universal, and sought to identify conditions where increased participation is most likely to improve outcomes (Birnbaum 2016; Bodin 2017; Newig et al. 2018). Box 1 lists some of the key points arising from this literature that are most applicable to chronic biological invasions:

Box 1: Key governance considerations for managing biological invasions

- **Conflicting objectives and allocating scarce resources:** more extensive and frequent dialogue, and clear decision-making processes are needed when stakeholders hold contrasting values (e.g. view invasive species as a pest vs a resource) or are competing for scarce resources (e.g. funding for management operations)

- **Complexity and the need for mutual learning**: complex problems (e.g. biological invasions driving multiple ecosystem impacts) require a range of knowledge sources and thus benefit from mutual learning across a diverse range of stakeholders

- **Spatial alignment of ecosystems and governance structures**: areas of interest for governance structures should correspond to ecosystems or regions insensitive to management decisions outside their boundaries (e.g. areas not vulnerable to colonisation from external invasive species populations).

- **Temporal alignment of governance structures and the problem**: long-term problems (e.g. chronic biological invasions) require governance structures that are sustainable in the long-term.

Adaptive governance: where science and values meet

We propose adaptive governance (governance systems which evolve in response to changes in understanding of or threats to ecological systems, economic conditions and stakeholder values, Folke et al. 2005) as a vehicle for interaction between science-generated forecasts and wilding conifer governance. Adaptive governance is specifically designed to deal with common resource pool problems, where limited resources (either environmental or financial) are shared across multiple stakeholders (Nelson et al. 2008). This is the case for management of chronic invasions, since funding is usually insufficient for total eradication, so that some form of prioritisation is required (Mason et al. 2016). Adaptive governance is strongly transdisciplinary, recognising that scientific information must be interpreted through social, spiritual and cultural values derived from multiple stakeholder perspectives (Karpouzoglou et al. 2016). The exact methodologies employed will vary through a process of experimentation within individual governance groups (Popa et al. 2015). However, a recent meta-analysis of 92 adaptive governance studies provides a range of good practice suggestions (Sharma-Wallace et al. 2018, Box 2).

Box 2: Good practices for successful adaptive governance

- **Foster diversity and manage power imbalances:** include representatives from each stakeholder or community group in decision-making forums
- **Establish formal processes:** for connecting relevant actors (e.g. local community groups, central and local government, scientists), coordinating governance activities (especially decision making), and disseminating information
- **Build social capital**: trust, familiarity, and goodwill underpin successful collaboration and governance outcomes. May require a long-term, sometimes contentious, collaboration process. Will benefit from informal collaboration opportunities, such as community events.
- **Facilitate community empowerment:** some form of local engagement is vital to successful implementation and maintenance of adaptive governance approaches.
- **Develop and maintain capacity:** access to scientific and non-scientific knowledge systems, mutual learning by stakeholders and researchers, sufficient resources for management actions, financial and organisational support to sustain governance activities.

Conclusions

This study demonstrates a practical framework for empowering biological invasions researchers to contribute their knowledge to the long-term, collaborative research efforts needed to manage chronic biological invasions. It also provides clear guidance for researchers in partnering with the governance bodies responsible for management decisions, and in assessing whether or not all relevant stakeholder groups are represented.

We have demonstrated that viewing knowledge ecologies as an integrated collection of distinct knowledge ecosystems offers a promising approach for identifying, generating, curating and integrating the knowledge sources needed to improve the management of chronic biological invasions. We have highlighted the importance of developing durable knowledge curation and generation platforms (i.e. knowledge ecosystems) for connecting research and management within chronic biological invasions. Knowledge ecosystems have the dual advantage of facilitating the contribution of researchers to a long-term, collective research effort, while also ensuring ongoing research support for decision-making by governance bodies. We have provided an example of knowledge ecosystem structure and function for invasive conifer *spread and impacts* here; knowledge ecosystems should also be built for the other knowledge sources required to address biological invasions (i.e. stakeholder values, management technologies and operational logistics).

Our knowledge ecology framework, by explicitly identifying high-level collaborative actions, provides practical foci for ongoing transdisciplinary collaborations spanning multiple knowledge sources and between researchers and non-researchers. This is especially true where science interacts with governance bodies to influence management decisions.

Finally, our study demonstrates the benefits of engaging with non-researchers for both the scientific robustness and relevance of research outputs. Thus, biological invasions researchers have much to gain through increased engagement with decision-making bodies responsible for the management of chronic biological invasions.

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Authors contribution NM developed the concepts, performed analyses and wrote the manuscript. OB and RP performed analyses. All authors commented on previous versions of the manuscript.

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Compliance with ethical standards

Conflicts of interest The authors declare that they have no conflict of interest.

Consent for publication This manuscript has been approved for submission by *Manaaki Whenua* Landcare Research.

Code availability All R script files used in the publication are available at: https://doi.org/10.7931/mwlr.datastore.

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