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The implications of climate change for biodiversity conservation and the National Reserve System: temperate grasslands and grassy woodlands

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Executive summary

Owing to widespread fragmentation and degradation associated with agricultural and other development, temperate grassy ecosystems (TGE), including grasslands and eucalypt woodlands, represent some of the most threatened ecological communities in Australia. The goals of this study were to predict how climate change may impact these ecological communities in south-eastern Australia, and develop recommendations for adapting conservation and management efforts towards better outcomes for biodiversity under climate change. We combined an expert knowledge approach (including a workshop with over 20 TGE experts) with two types of community-level spatial modelling (using artificial neural networks, ANN, and generalised dissimilarity modelling, GDM) to inform management responses. We focused on the following questions: (1) What are the structuring ecological processes in TGE? (2) How are 'biotically scaled' environments likely to be impacted by climate change? (3) What attributes of TGE could influence the response of biodiversity to altered environments and processes? (4) What major ecological changes are likely? (5) What conservation management options are available to address these?

The modelling approaches used in this study attempted to predict the 'biotically scaled' stress that the TGE biome will be subject to under different climate scenarios. ANN models, focusing on vegetation structure, suggest very high mean stress in woodlands and grasslands across the biome, under medium and especially high 2070 climate scenarios. Spatially, predicted ANN stress is more moderate on the New South Wales north-west slopes, and tends to increase southwards and inland. However, interpretation of these results should be tempered by the fact that ANN predicted moderate stress even for the current climate, suggesting a relatively high degree of overlap in environments of these structural types. Indeed, around 34% of current TGE environments are predicted by ANN to favour other structural types, particularly *Callitris*, *Casuarina* or *Eucalyptus* forest or chenopod shrublands along the inland and eastern margins. Under medium 2070 scenarios, environments favouring eucalypt woodland persist predominantly on the New South Wales north-west slopes and along narrow margins of the Great Dividing Range, with small expansions in coastal areas such as the Gippsland Plain. These patterns are accentuated under 2070 high scenarios. Where environments no longer favour eucalypt woodlands, environments within the current TGE biome are predicted to become increasingly favourable for chenopod shrublands from inland edges (and even parts of Tasmania where this structural type does not currently exist), with patchy areas favouring *Casuarina* forest environments. A number of other structural formations are potentially favoured in more mesic areas. Notably, the extent of environments favouring grasslands is predicted to increase. This is despite very high predicted ANN stress in current grasslands, reflecting a changing spatial location of these environments.

GDM produced estimates of stress based on expected compositional dissimilarity under the present vs. future climate scenarios for each 1 km^2 grid cell. These showed similar patterns in grasslands and woodlands, with high stress or compositional dissimilarity predicted for reptiles, snails and plants, but low to moderate stress for birds, mammals and frogs. Lower estimates for the latter possibly reflect the wider distributions of species in these groups, but we emphasise that models did not account for changing habitat characteristics such as might be experienced with high levels of plant compositional dissimilarity. For nearly all groups, predicted stress is at the lower end of the range in Tasmania. For mammals, reptiles, snails and plants there is a tendency for lower stress in central-west Victoria and higher stress in central New South Wales, and sometimes South Australia. The New South Wales north-west slopes, which appear least stressed in the context of woodland structural formations (ANN), usually show lower stress than central New South Wales but do not stand out ahead of Tasmania or central-west Victoria. Birds and frogs show a somewhat different pattern, with stress tending to increase from the inland edges of TGE towards the more coastal edges.

Although models predict high environmental stress within the TGE biome under climate change, outcomes for TGE will be mediated by a range of attributes of component individuals, populations, species, ecosystems and landscapes that confer resilience to changing environments. Prior to degradation, a range

of TGE characteristics offered mechanisms for effective resistance or adaptation to environmental change, including high levels of genetic diversity and large population sizes in widespread species, high levels of species diversity and redundancy, and moderate environmental heterogeneity and connectivity to permit adaptation through local-scale migrations. On the other hand, other attributes will constrain the capacity of TGE to adapt. Few TGE native plant species have long-lived seed banks and many are poor dispersers, limiting their capacity to persist or migrate. Many TGE fauna are dependent on tenuous biotic interactions, such as the temporal connectivity of resources, that will increase their vulnerability. Further, TGE attributes such as large population sizes and high genetic diversity, species diversity and connectivity that would normally confer resilience have been highly compromised through agricultural and other development. Similarly, degradation of site-scale ecosystem processes, such as soil water infiltration and habitat structure, will affect the capacity of ecosystems to cope with the additional stresses of climate change, and interactions with exotics may facilitate community reassembly by exotic rather than native species.

Based on model predictions and expert understanding of driving ecological processes in TGE (particularly climate, soils and fire), five potential outcomes of climate change in TGE are discussed, acknowledging that they are subject to substantial uncertainty. These include potential changes in fire frequency; structural change, including declining tree cover and altered shrub-grass balance; shifts in the functional composition of the herbaceous ground layer, such as changed C_3/C_4 and perennial/annual ratios; predominance of exotic species such as buffel grass (*Pennisetum ciliare* [syn.] *Cenchrus ciliaris*) in novel communities; and cascading changes in ecological interactions. We emphasise that, while it is not likely to be possible or desirable to entirely prevent ecological change in TGE, it is possible to enhance resilience of natural systems by ameliorating non-climatic limits to adaptation, and to promote reassembly by native rather than exotic species. Current conservation and restoration efforts already target similar goals, and reinforcing and adjusting these approaches offer the highest priority, lowest risk climate adaptation options in the degraded TGE biome. Conservation options discussed in this context include:

(1) *Continue to conserve suites of native species in fragmented TGE landscapes through appropriate management and protection of a diversity of sites, representing natural environments and processes across the biome.* Given the highly degraded and fragmented state of this biome, achieving a comprehensive, adequate and representative reserve system in TGE is challenging even in the absence of climate change, so continuing to protect remnant vegetation is of high priority. Although all species may not persist, these sites are likely to be the best local source of native species from which novel ecosystems may reassemble, and will provide the most appropriate sites and environments in which this can occur. Under climate change a range of priorities become increasingly relevant, in particular, sites with natural (non-enriched) soil environments that will facilitate reassembly by native rather than exotic species, and areas predicted to experience lower biotically scaled stress ('refugia').

(2) *Tailor current restoration efforts and carbon plantings towards climate resilient outcomes.* Current activity in restoration and revegetation offers significant opportunity to experiment with novel approaches to climate adaptation that might be considered too great a risk to extant native diversity in more intact landscapes. We suggest investigating options for (a) planting more diverse mixes of local species to increase the chance that some will survive, (b) using genes and species reflecting the degree of expected change (e.g. augmenting genetic diversity of local populations with genotypes potentially more suited to predicted climates), (c) varying levels of connectivity to balance the benefits of native species migrations with the disadvantages of invasions, and (d) reducing non-climatic stressors such as degraded soil physical conditions.

(3) *Favour land use changes with potential for positive rather than negative biodiversity outcomes*. Current and future land use changes – driven by changing viability of grazing and cropping enterprises, urbanisation, and emergence of new markets for carbon and biofuels – could ameliorate or magnify climate change impacts. Revegetation, carbon sequestration initiatives, abandonment and reversion to low-input native pastures offer potential advantages for TGE biota. By contrast, replacement of low intensity land uses with more intensive agriculture, such as irrigated vineyards or biofuel crops will exacerbate the negative outcomes of climate change for biodiversity. As well, plantings for carbon sequestration could

favour woody rather than grassy communities. Policies attempting to moderate land use change will thus need to balance the social and economic benefits with biodiversity impacts.

(4) Manage non-Australian exotics at a range of levels to promote reassembly by native species.
Novel communities dominated by exotic species are a likely negative outcome of climate change, hence strengthening of current approaches to exotic management is needed to minimise these impacts. These include quarantine, identification and control of sleeper (incipient) exotics, containment of incipient transformer exotics (particularly buffel grass), as well as more intensive weed and feral control in selected conservation assets.

(5) *Manage disturbance, particularly fire, to maintain open grassy ecosystems at selected locations.* Given the general human desirability of open woodland landscapes and the poor outlook for TGE elsewhere in south-eastern Australia, it may be desirable to manage some areas to promote persistence of the open, grassy structure of TGE. Shrub–grass dynamics in semi-arid ecosystems are relatively well-understood, and it is likely that they can be successfully manipulated using fire and grazing.

1 Introduction

Accelerated climate change associated with the greenhouse effect will have dramatic consequences for Australia's ecosystems (Fischlin and Midgley 2007; Dunlop and Brown 2008). Irrespective of mitigation efforts, past emissions of greenhouse gases will contribute to unavoidable climate change for more than a century (Fischlin and Midgley 2007), and there is already compelling evidence for altered species ranges and changed phenologies around the globe (Thuiller et al. 2008; Steffen et al. 2009). Consequently, strategies need to be incorporated into conservation planning and management to maximise the resilience of natural systems to climate change (Dunlop and Brown 2008; Heller and Zavaleta 2009). Towards this goal, many studies aim to predict changes in species distributions under future climate scenarios (Thuiller et al. 2008; Yates et al. 2010). Few studies in Australia have attempted to understand how climate change might impact on terrestrial ecosystems, landscapes or ecological communities (but see Hilbert et al. 2001; Williams et al. 2007a; Dunlop and Brown 2008; Hovenden et al. 2008; Steffen et al. 2009; Hovenden and Williams 2010), yet these are the scales at which conservation management most frequently occurs (Ferrier and Guisan 2006).

Spatial modelling of biodiversity is increasingly applied at the ecological community level, using a range of approaches (Ferrier and Guisan 2006). These methods offer an opportunity to move beyond single-species approaches to predicting broad biodiversity impacts of climate change (e.g. Hilbert and Ostendorf 2001; Dirnböck et al. 2003; Peppler-Lisbach and Schröder 2004). In particular, they can indicate levels of environmental stress extant ecosystems are likely to be subject to, as well as expected distributional shifts of their current realised environments. Notwithstanding these benefits, extrapolation of models to new environments is inherently risky for many reasons, including lack of model-training data for novel combinations of environment variables, inadequate accounting for ecological and evolutionary processes, and uncertainties in climate models (Ferrier and Guisan 2006; Thuiller et al. 2008; Elith and Leathwick 2009). Their effective translation to conservation and management applications thus requires cautious interpretation in association with expert ecological knowledge of the systems of interest.

Temperate grasslands and grassy woodlands (hereafter temperate grassy ecosystems, TGE) are widespread in the transitional zone between the higher rainfall forested margins of Australia and the shrub woodlands, shrublands and hummock grasslands of the drier interior (DEWR 2007). They support a characteristic and diverse flora and fauna, and represent the iconic Australian landscapes of early Australian artists (Prober and Hobbs 2008). However, because they predominate on productive agricultural land, they are some of the most poorly conserved vegetation communities in Australia (DEWR 2007). Clearing, fertilisation, inappropriate grazing and cultivation have led to dramatic losses of plant and animal diversity and have compromised ecological and hydrological processes, resulting in the decline of remnant trees; salinisation, erosion and acidification of productive land; and widespread invasion by weeds and feral animals (Prober and Thiele 2005).

With a goal of ameliorating this widespread degradation, considerable conservation effort in the past 20 years has focused on reservation, ecological restoration and revegetation in the TGE (Prober and Thiele 2005). Despite this, there has been minimal research on likely impacts of climate change on biodiversity in these communities (but see Williams et al. 2007a; Hovenden et al. 2008; Hovenden and Williams 2010), and few attempts to tailor conservation efforts towards climate-resilient outcomes. The goals of this report were to predict how climate change may impact these ecological communities, and to develop recommendations for adapting conservation and management efforts towards better outcomes for biodiversity under climate change.

Towards this goal, we combined an expert knowledge approach with two types of community-level spatial modelling (using artificial neural networks, ANN, and generalised dissimilarity modelling, GDM) to inform management responses. We focused on the following questions: (1) What are the structuring ecological patterns and processes in the target ecosystems? (2) How are 'biotically scaled' environments likely to be impacted by climate change? (3) What attributes of the target systems could influence the response of biodiversity to altered environments and processes? (4) What major ecological changes are likely? (5) What conservation management options are available to address these? Our emphasis was on biophysical change, but the social dimensions of change were considered where they might ameliorate or exacerbate climate change impacts on biodiversity (see also Dunlop and Brown 2008).

2 Methods

For the purposes of this study, temperate grassy ecosystems (TGE) were defined according to vegetation classification and mapping collated by the National Vegetation Information System (NVIS) (DEWR 2007). We included the major vegetation subgroups ‘eucalypt woodlands with a grassy understorey’ (MVS 9), ‘temperate tussock grasslands’ (MVS 36) and ‘other tussock grasslands’ (MVS 37) to best represent TGE, acknowledging that the NVIS data are imperfect and include inconsistencies from state to state. The NVIS classification is an Australia-wide system, so to capture TGE of south-eastern Australia (in contrast to subtropical and tropical grassy ecosystems) we confined our analysis to areas south of 27.5˚S (around Toowoomba, Queensland) and east of 135.5˚E (Eyre Peninsula, South Australia, Figure 1).

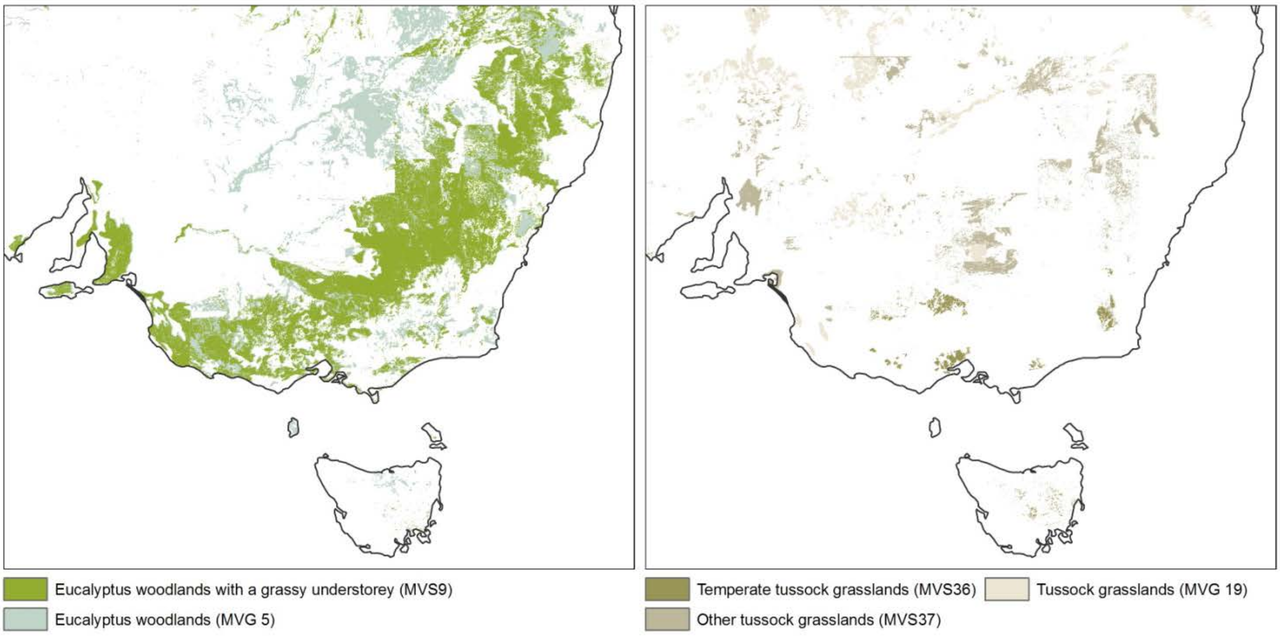

Figure 1 Distribution of major vegetation subgroups (MVS) used to define the TGE biome. For reference to ANN models, pale blue shows additional (non-grassy) eucalypt woodland contributing to major vegetation group (MVG) 5 (eucalypt woodland); pale brown shows additional grassland (Mitchell grasslands and Blue grass grasslands) contributing to MVG 19 (tussock grassland), that were not included in our definition of TGE

To explore likely climate change impacts and potential management responses in TGE, we reviewed the ecological literature on the biome and drew from a workshop involving a panel of more than 20 TGE experts from 14 different conservation research and management institutions representing the five Australian states in which the biome occurs (Appendix 1). Expertise included woodland and grassland plant ecology, vertebrate and invertebrate ecology, fire ecology, conservation and management planning and implementation, ecological modelling, and National Reserve System (NRS) policy and practice. Workshops were structured around the following questions:

1. What environmental and ecological characteristics make the TGE biome distinctive?
2. What types of changes are likely to affect TGE, its biota and socio-economic characteristics?
3. What are the implications of different kinds of changes for biodiversity and the NRS?
4. How might the biome and NRS, in a whole-landscape context, be managed differently under climate change?

We augmented this expert knowledge approach with two modelling approaches to estimate the extent of 'biotically significant' stress the biome might be subject to under specific climate scenarios: (1) modelling of the ecological environments of vegetation communities classified on the basis of vegetation structure, using artificial neural networks (hereafter ANN, Hilbert and Van Den Muyzenberg 1999), and (2) simultaneous modelling of community composition and associated environments, using generalised dissimilarity modelling (hereafter GDM, Ferrier and Guisan 2006).

2.1 Climate scenarios

Uncertainty about future greenhouse gas emissions and limitations of global climate models contribute to high levels of uncertainty in future climate projections worldwide. Consequently, realistic climate adaptation management must occur in an uncertain environment (Hagerman et al. 2010). The expert panel thus considered broad regional climate projections derived from ensembles of global climate models and multiple greenhouse gas emission scenarios (CSIRO 2007). These suggested that by 2070 there would be general warming in all seasons by 1–6 °C (mid-point 1.5–2.5 °C) across the biome. Predictions for change in rainfall by 2070 are particularly variable, ranging from -40% to +10% annually (mid-point -20% to -5%), with potential reductions of up to 60% in winter and spring, and potential change of -40% to +40% in summer and autumn.

Modelling approaches addressed two more specific climate scenarios, both using outputs from the CSIRO Mk3.5 GCM downloaded from OzClim (CSIRO 2012): a *medium* impact scenario, using the A1B emissions scenario; and a *high* impact scenario, using the A1FI emissions scenario (IPCC 2000). Monthly climate change grids were downloaded at 0.25° resolution for maximum temperature, minimum temperature, rainfall and evaporation, by specifying the above scenarios in OzClim. Spatial downscaling was carried out using the ANUCLIM software (Houlder et al. 2000; Fenner School of Environment and Society 2012), which incorporates three submodels: ESOCLIM, which outputs raw climate variable grids; BIOCLIM (Busby 1986), which outputs grids of bioclimatic parameters; and GROCLIM, which can output gridded indices from simple growth models. The beta release of ANUCLIM version 6.0 was used, which allows climate change grids to be applied over the historical 1990-centred climate surfaces. Software (Harwood and Williams) was written to interpolate the raw 0.25° CSIRO grids to cover the whole Australian landmass, and relate evaporation change to the date range used in ANUCLIM 6. Following this interpolation, monthly maximum temperature, minimum temperature, rainfall, and evaporation change grids were input into ANUCLIM 6 with a 0.01° digital elevation model. The result was a suite of monthly 0.01° (≈1 km^2) resolution future climate surfaces for maximum temperature, minimum temperature, rainfall, evaporation and radiation, with 35 BIOCLIM variables and four plant growth indices for each scenario.

2.2 Classification of environments using artificial neural networks

This approach used maps of vegetation classes along with detailed, spatial estimates of climate, topographic and edaphic variables to objectively classify environments that are characteristic of these vegetation classes. The goal was to transform a high dimensional, physical environment space (many climate variables and a number of terrain and soil variables as well) into a lower dimensional, ecologically meaningful space. This was accomplished through supervised classification. Then, given any spatial scenario of change in the climate we can map these ecological environments in geographic space. Most importantly, we can compare this new spatial map of environments with what we estimate it is today and also with the spatial distribution and extent of the actual ecological classes. In this way, we can quantify how the extent and distribution of the environmental classes may change in the future and infer how climate change may affect vegetation classes and, consequently, biodiversity and function.

We used ANNs for the supervised classification of environments based on mapped vegetation classes. This methodology builds on the successes of a similar approach that was used in the Wet Tropics Bioregion of north-east Queensland where an ANN was used to classify 15 structural/physiognomic forest environments based on a range of climatic, edaphic and topographic variables (Hilbert and Van Den Muyzenburg 1999; Hilbert et al. 2001). For all of Australia, we classified environments for the 23 major vegetation groups (MVGs) described in DEWR (2007) (note this is a higher level of classification than major vegetation sub-groups (MVSs) that we used to define TGE, see Figure 1). The MVG data consist of a digital map of their pre-clearing distributions at a one hectare resolution for the entire continent (Thackway et al. 2007). The major vegetation subgroups defining TGE are subgroups of two MVGs 'eucalypt woodlands' (MVG 5) and 'tussock grasslands' (MVG 19); ANN modelling could only be applied at this more general level. ANN models thus need to be viewed as indicating broader environments than currently occupied by temperate TGE, and in particular should be interpreted in terms of broad structural formation rather than species composition.

We used FANN (Fast Artificial Neural Network Library) to classify environments of the MVGs. This software is an open source neural network library available from http://leenissen.dk/fann/, which implements multilayer artificial neural networks in the programming language C. We used a single, multiple-output neural network to classify the available environmental variables by MVG class with 35 input nodes (23 bioclimatic variables, three soil variables and nine topographic variables), 150 hidden nodes and 23 output nodes representing the MVGs. We used the largest output node value to map MVGs in the current and climate change scenarios. We also trained individual classifications for each of the MVGs using the Tiberius software (Brierley unpublished) to rank variable importance using the Gini Coefficient (Breiman et al. 1984).

The ANNs provide much more information than is apparent in a classification, where the output node with the largest value is chosen as a pattern's (location's) classification. By using the values of all the output nodes we calculated the dissimilarity of this vector to the vector with the value of 1.0 for the class that is mapped at that location and all other values of 0.0. The dissimilarity is the vector angle between the two, normalised to the range [0,1] (Hilbert and Van Den Muyzenburg 1999). For example, a location that is mapped as Rainforest and vine thickets with a dissimilarity of 0.1 has an environment that is more typical of this class than another location, also mapped as this class, with a dissimilarity of 0.4. Hilbert and colleagues (Hilbert et al. 2001; Hilbert and Ostendorf 2001) interpret dissimilarity as an index of relative environmental stress. It could also be thought of as a propensity to change. Dissimilarities greater than 0.5 indicate environments that are more like that of some other class than the one that is mapped.

A more detailed description of the ANN methods used in this project in the accompanying report, Hilbert and Fletcher (2012).

2.3 Analysis of biotically scaled environmental stress using generalised dissimilarity modelling

Generalised dissimilarity modelling (GDM) is a statistical technique for modelling the compositional dissimilarity between pairs of geographical locations, for a given biological group (e.g. reptiles), as a function of environmental differences between these locations (Ferrier 2002; Ferrier et al. 2002, 2007). The measure of compositional dissimilarity (*d*) employed in this project is the Sorenson, or Bray–Curtis, index:

$$d_{ij} = 1 - \frac{2A}{2A + B + C}$$

where *A* is the number of species common to both locations *i* and *j*

B is the number of species present only at location *i*

C is the number of species present only at location *j*

In other words, based on this measure, the compositional dissimilarity between a given pair of locations is the proportion of species occurring at one location that do not occur at the other location (averaged across the two locations) – ranging from 0 if the two locations have exactly the same species through to 1 if they have no species in common.

GDM uses data on species recorded at a sample of locations across the region of interest to fit a model predicting the compositional dissimilarity between pairs of locations as a non-linear multivariate function of the environmental attributes of these locations. Another way of viewing this is that GDM effectively weights and transforms the environmental variables of interest such that distances between locations in this transformed multidimensional environmental space now correlate, as closely as possible, with observed compositional dissimilarities between these same locations (see Ferrier et al. 2007 for full explanation).

This project employed a set of GDM models already derived for the Australian continent by a separate (then) DEWHA-funded Caring for Our Country Open Grants project performed by CSIRO in collaboration with DEWHA and the ANU Fenner School of Environment and Society (Williams et al. 2010). These models were derived using continent-wide biological data collated within DEWHA's Australian Natural Heritage Assessment Tool (ANHAT) database – a compilation of species-location records from a large number of herbaria, museums, State and Commonwealth departments, and private individuals. The models were fitted at 1 km^2 grid resolution[1] across the entire continent using best-available environmental layers for 76 climate, terrain and substrate variables (Williams et al. 2010). Models were derived for 12 different biological groups, six of which were employed in the work described in this current report:

- vascular plants (model based on data for 12,881 species at 374,640 locations – i.e. 1 km^2 grid cells)
- land snails (model based on 2,774 species at 19,118 locations)
- frogs (model based on 218 species at 100,143 locations)
- reptiles (model based on 819 species at 83,661 locations)
- birds (model based on 690 species at 242,814 locations)
- mammals (model based on 298 species at 100,369 locations).

The current project used the above models to infer potential changes in biological composition as a function of projected changes in climate across the continent. This is based on the assumption that the amount of change in species composition expected for location *A* as a result of climate change, will be equivalent to the compositional dissimilarity currently observed between location *A* and another location *B* with a current climate matching that projected for location *A* (Ferrier and Guisan 2006, Ferrier et al. 2007). It is likely that the actual change in biological composition resulting from climate change will be shaped by many factors, and associated sources of uncertainty, beyond those considered in this modelling – e.g. biotic interactions, indirect effects of changed fire regimes, dispersal ability, lag effects, adaptation capacity and plasticity. The level of compositional change predicted by the GDM approach is therefore best interpreted as no more than a relative indicator of potential 'environmental stress' expected to be experienced by species in a given biological group under a given climate scenario.

The GDM-based analyses performed in this project resulted in statistics and maps associated with the following (terms in italics concur with terms used in results for each measure):

- The predicted *'point scale' dissimilarity* between the current composition of each grid cell and its composition under a given climate scenario, as a general indicator of potential environmental *stress* on a cell-by-cell basis. This was estimated and mapped separately for each of the six biological groups (listed above). All of the remaining analyses below were performed for vascular plants only.
- The *absolute minimum dissimilarity* between the current composition of each cell and the predicted future composition of all cells on the continent under a given climate scenario, as an indicator of 'disappearing [biotically scaled] environments' (as per Williams et al. 2007b).
- The minimum predicted dissimilarity between the future composition of each cell under a given scenario and the current composition of all cells on the continent, as an indicator of '*novel environments*' or 'no-analogue [biotically scaled] environments' (as per Williams et al. 2007b).
- Two measures of the potential contribution that environmental heterogeneity around each cell may make to ameliorating the effects of a given climate scenario:

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[1] The models were fitted to data based on 0.01° by 0.01° grids, which are approximately 1 km by 1 km, but their exact dimensions vary with latitude.

    a. the proportional *change in effective habitat area* within a surrounding radius varying from 750 m up to 100 km, where ‘effective habitat area’ is the summed area of all cells within this radius, with each cell weighted according to the predicted similarity (1-d_{ij}) between the composition of this cell (current versus future) and the current composition of the focal cell (see Ferrier et al. 2004, and Allnutt et al. 2008 for a more detailed explanation of this concept)
    b. the predicted dissimilarity between the current and future composition of each cell (from the first dot-point above), minus the minimum predicted dissimilarity between the current composition of this cell and the future composition of any other cell within a radius varying from 750 m to 100 km, indicating the *buffering* provided within that radius.

- An extension of the analysis of *change in effective habitat area* described above to consider the added effect of habitat loss and fragmentation. In this case only cells mapped as extant vegetation (based on the NVIS extant vegetation layer) were allowed to contribute to the calculation of effective habitat area. We acknowledge that this analysis is subject to the constraints of inaccuracies in the NVIS data layer.

A more detailed description of the GDM methods is provided in the accompanying report by Ferrier et al. (2012).

3 What are the structuring ecological processes in TGE?

Consistent with key drivers of plant distributions worldwide, climate, soils and disturbance are the major drivers of ecosystem functioning and regional variation in TGE. In woodlands, trees also contribute significantly to many ecosystem functions.

The biome is characterised by cold winters and mild to hot summers. Plant composition varies gradually across north–south gradients in temperature and rainfall seasonality, and more steeply with rainfall gradients from around 400 mm mean annual rainfall in the interior to 800+ mm in more coastal regions (Kirkpatrick et al. 1995; Prober 1996; Keith 2002; Prober and Thiele 2004; BoM 2009). Peak plant growth is dependent on rainfall seasonality, ranging from winter–spring dominated in the cooler southern zones, towards more summer-dominated in the north, with limits to growth being cool winter temperatures and/or dry summers (Hobbs and Yates 2000; Hutchinson et al. 2005). Reflecting this, the native grassy ground layer often includes a mix of cool-season to year-long green C_3 perennial grasses (especially *Poa, Austrodanthonia* and *Austrostipa* species) and warm-season C_4 perennial grasses (particularly *Themeda triandra, Enteropogon* and *Bothriochloa* species), permitting active growth in response to rainfall in all but the coldest times of year (e.g. Moore 1970; Lodge and Whalley 1989). Notably though, much of the native plant diversity of the biome is contributed by cool-season to year-long herbaceous forbs and graminoids. These species are typically perennial, but there is an increasing component of annual forbs as mean annual rainfall declines (Whalley et al. 1978; Prober and Thiele 2004).

Unlike much of Australia, soils of TGE are naturally deep, often high in clay and moderate to rich in most nutrients, important for supporting the dense, herbaceous ground layer (Beadle 1981; Kirkpatrick et al. 1995; McIntyre et al. 2002; Prober et al. 2002a; Martin 2003). The ground layer in turn protects soil surfaces from erosion and exposure to sunlight and raindrop impact, facilitating optimal moisture infiltration, biological activity and nutrient cycling (Prober et al. 2008). The foraging action of small ground-dwelling mammals (Martin 2003), and in more open, arid or recently burnt environments, a surface crust of non-vascular plants (bryophytes, lichens and algae; Eldridge 2001) can similarly contribute to the maintenance of these soil processes. Importantly, while otherwise fertile, native topsoils of TGE are usually naturally low in available macronutrients (nitrogen, phosphorus and sulphur; Morgan 1998; Prober et al. 2002a, 2002b). This is likely to underlie the negative response of many native species to nutrient enrichment associated with soil disturbance or fertilisation in TGE (see below).

Although generally fertile, TGE occupy a broad range of soil types, from alluvial, aeolian, igneous, sedimentary and metamorphic origins, and these contribute to differentiation among woodland and grassland types (Beadle 1981; Benson 2008). In particular, heavy, waterlogged and/or cracking clay soils can restrict tree recruitment, so commonly delimit boundaries between grassland and woodland (Beadle 1981; Kirkpatrick et al. 1995). On the other hand, soils can vary considerably within woodland types with only subtle variations in floristic composition. These typically involve increasing grass dominance, and declining subsidiary plant diversity and shrub abundance, with increasing soil fertility or in lower topographic positions (Lunt and Bennett 2000; Prober and Thiele 2005). Trees are another important driver of small-scale soil heterogeneity, acting as wicks for water infiltration (Eldridge and Freudenberger 2005), concentrating nutrients beneath their canopies, and influencing understorey composition (Prober et al. 2002a).

Disturbances such as marsupial grazing, fire and soil disturbance by ground-dwelling mammals and Aboriginal Australians played an important role in the dynamics of TGE prior to European settlement (e.g. Gott 1983, Kirkpatrick et al. 1995, Martin 2003, Lunt et al. 2012). Owing to the near extinction of ground-dwelling mammals and the cessation of Aboriginal people digging for food, impacts of these soil disturbances are poorly known (Martin 2003). While generally poorly characterised, fire and marsupial grazing have been shown to influence plant recruitment and composition, including the dominant woodland trees (Vesk and Dorrough 2006), shrubs (Knox and Clarke 2006; Watson et al. 2009) and herbaceous species (Morgan 2001; Prober et al. 2007). In particular, it is likely that fire contributed to maintenance of the open grassy structure of many TGE, and to the balance of C_3 vs C_4 grasses (Lunt et al. 2012). In productive TGE (i.e. areas with higher rainfall and fertile soils) disturbance can promote subsidiary forbs by reducing the dominance of robust C_4 tussock grasses (especially *Themeda triandra*), but the resilience of grassy ecosystems to frequent fire is lower in less productive (drier or infertile) ecosystems (Lunt et al. 2012).

Faunal assemblages of TGE are strongly associated with climate and vegetation structure. Often, they overlap with assemblages of adjacent vegetation types, but a substantial suite of species, including a distinctive group of woodland-dependent birds and grassland dependent reptiles, is centred on the TGE biome (Robinson and Traill 1996; Lunt and Bennett 2000). Woodland-dependent birds have evolved in response to the long growing season and year-round availability of food, combined with the characteristic structure of the woodland habitat. The open canopy of large-crowned trees supports many obligate hollow, canopy and bark-dependent fauna such as parrots, cockatoos, gliders and many invertebrates, and provides a significant source of nectar and other carbohydrates (Robinson and Traill 1996; Lunt and Bennett 2000). Some species prefer treeless habitats; for example, the endangered Plains-wanderer (*Pedionomus torquatus*) is restricted to open grassland habitats in south-eastern Australia (Baker-Gabb 1990). The relatively mild climate and year-round food supply of TGE also make them an important destination for migratory birds (Lunt and Bennett 2000).

4 Threats and changes since European settlement

Owing to the productive landscapes they occur on, TGE have been widely modified through agricultural and other development, leading to their classification as some of the most threatened ecological communities in Australia (Prober and Thiele 2005; DEWR 2007). Based on DEWR (2007), TGE once collectively covered 379,000 km^2 in south-eastern Australia, over 80% of which was woodland (Table 1). However, most of these areas have been cleared and the remaining woodlands and grasslands are typically modified or degraded (Figure 2a, Figure 2b). Only 2.2% are reserved within the NRS (Table 1, Figure 2c, Figure 2d).

Table 1 Mapped area of each major vegetation subgroup of the TGE biome within the south-eastern Australian region defined in this study (using pre-1750 extents in DEWR 2007), compared with area in the National Reserve System

MVS	BIOME AREA (KM^2)	NRS AREA (KM^2)	PERCENT IN NRS
MVS9 (grassy woodland)	312 057	6 778	2.17%
MVS36 (grassland)	9 974	245	2.46%
MVS37 (grassland)	56 770	1 235	2.18%
Total	378 801	8 258	2.18%

The current state of TGE has resulted from interactions among a number of primary human interventions that began with European settlement in the early nineteenth century, including widespread vegetation clearing for agriculture and infrastructure development, altered fire management, livestock grazing, exotic species introductions, and soil cultivation and fertilisation (Hobbs and Yates 2000; Dorrough et al. 2004; Prober and Thiele 2005; Lunt et al. 2007; Dorrough and Scroggie 2008). These degrading processes have impinged directly on biodiversity in TGE across much of their range and, importantly, continue to influence biodiversity indirectly through their complex effects on underlying ecosystem processes.

These indirect effects can limit the recovery of degraded TGE, so that simple removal of a primary degrading process (e.g. livestock grazing) is often insufficient for restoring woodland biodiversity (Prober and Thiele 2005; McIntyre and Lavorel 2007). In particular, loss of landscape connectivity and temporal resource continuity impact on the persistence of native birds and mammals (e.g. Radford et al. 2005); altered hydrological processes lead to salinisation (Cramer and Hobbs 2002); and soil nutrient enrichment promotes weed invasion, disadvantages many oligotrophic forbs and prevents native plant establishment (Prober et al. 2002b; Vesk and Dorrough 2006; Smallbone et al. 2007; Dorrough and Scroggie 2008). Further, introduction and widespread establishment of feral animals such as the European fox (*Vulpes vulpes*) has seriously impacted on the viability of populations of many small vertebrates already compromised through habitat loss (Martin 2003). Finally, drivers of fire ignition and spread have changed dramatically due to lack of burning by Aboriginal people, fire suppression, landscape fragmentation and livestock grazing. These have resulted in long fire intervals in many remnant TGE (Lunt et al. 2012), and recent studies suggest this may already be resulting in widespread invasion by woody natives such as *Callitris* spp. in wetter parts of its range (J. Cohn unpub. data).

At the landscape scale, agricultural development and associated changes to ecosystem processes have not affected all grassy ecosystems equally. Communities of the most productive lands were preferentially cleared and utilised (Hobbs and Yates 2000), leading to severe vegetation fragmentation with often <10% remnant vegetation (Hobbs and McIntyre 2005). In regions with less extensive cropping and pasture improvement, landscape changes are more diffuse, leading to variegated landscapes that support more native species within the general landscape matrix, for example, in native pastures (McIntyre et al. 2002). Sub-alpine woodlands and grasslands are typically in better condition; while they suffered significant degradation owing to summer grazing practices, they were not widely cleared, and most livestock grazing has ceased in the past 50 years (Williams and Costin 1994). Although much of the clearing and land use change in agricultural regions occurred over 50 years ago, continued clearing, loss of paddock trees and both increasing and decreasing land use intensity continue to contribute to significant landscape change today (Robinson and Traill 1996; Gibbons and Boak 2002).

5 Current conservation approaches

Given the highly degraded and fragmented state of this biome (Figure 2a, Figure 2b), achieving a comprehensive, adequate and representative (CAR) reserve system in TGE is challenging even in the absence of climate change. Few large, diverse areas remain available for reservation, as evident in the low percentage of this biome in the NRS (Table 1) and the small and scattered nature of NRS reserves in this biome compared with others in the region (Figure 2c, Figure 2d). Rather, flora and fauna are dependent on suites of vegetation remnants in varying conditions and under different land tenures. These include significant public lands held as Travelling Stock Routes, State Forests, town commons and reserves, as well as privately owned land. Conservation needs in these ecosystems thus need to be met through a mix of on- and off-reserve approaches (Kirkpatrick et al. 1995; Lambert et al. 2000; Thiele and Prober 2000), in cooperation with many landholders and land managers.

Conservation effort in TGE currently includes three main on-ground approaches (Lambert et al. 2000; Thiele and Prober 2000):

1. protection, optimal management and restoration of remnant vegetation patches and native pastures, to conserve and enhance site-scale conservation values
2. revegetation and other initiatives to increase habitat area and support critical ecosystem processes at the landscape scale
3. coordinated programs to limit the abundance and spread of exotic.

From an implementation perspective, a range of policy instruments and support programs aim to facilitate these on-ground outcomes, and to minimise further degradation through clearing or increased land use intensity. These include legislative regulation (e.g. clearing controls and threatened ecological community listings); legal protection through property rights (covenanting programs, formal reservation); and education and incentives programs to encourage community conservation (e.g. the Australian Government Caring for our Country box-gum grassy woodland stewardship scheme, the Tasmanian Government's Non-Forest Vegetation Project) (Lambert et al. 2000; Thiele and Prober 2000). Bodies such as Conservation Management Networks and regional Natural Resource Management councils have been established to facilitate the coordination of activities (e.g. Prober et al. 2001).

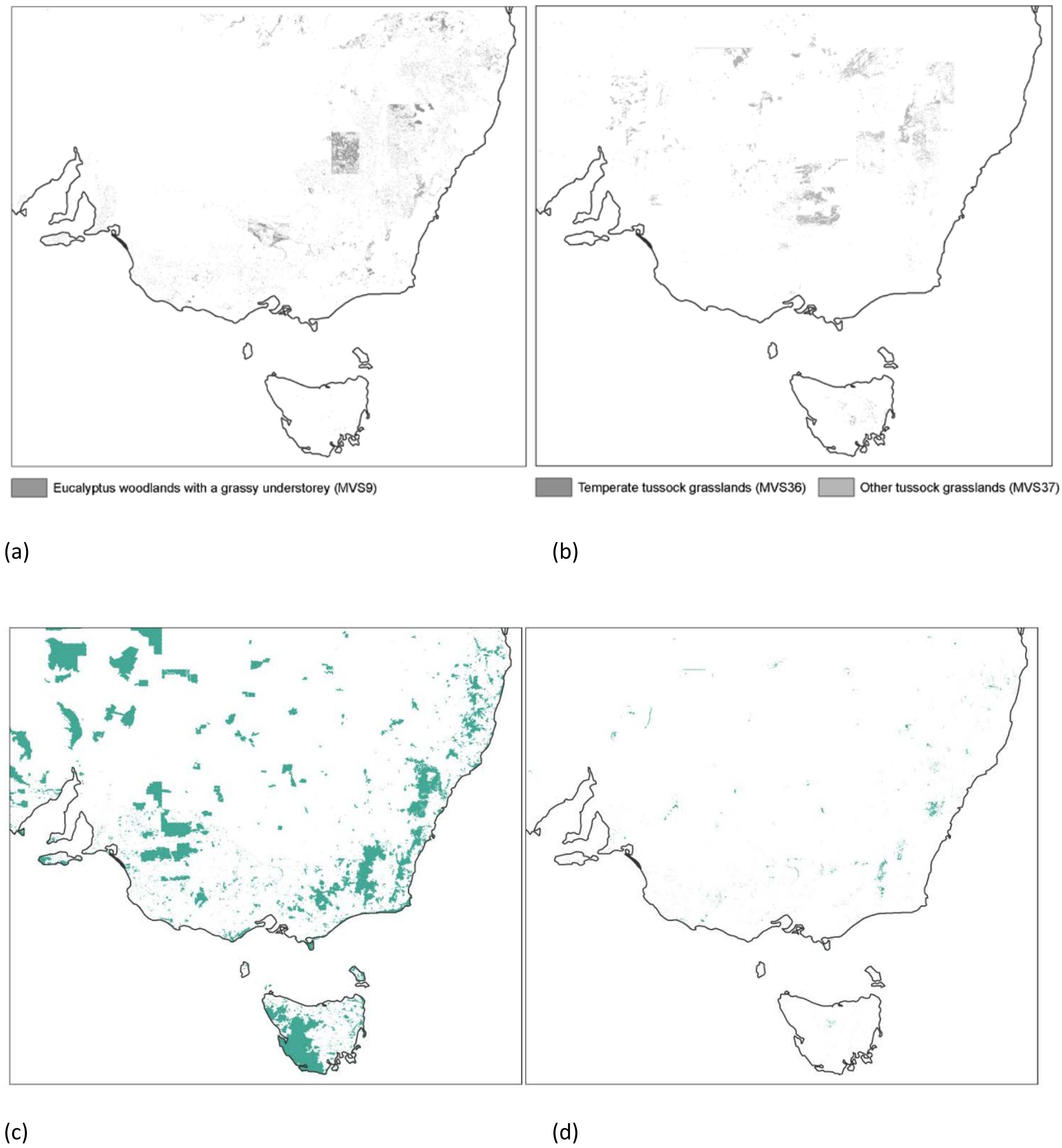


Figure 2 Extant vegetation in (a) MVS 9 and (b) MVS 36 and MVS 37 (sourced from DEWR 2007); and NRS protected areas (green, CAPAD 2006) within (c) the region and (d) the biome as defined by MVS 9, MVS 36 and MVS 37

6 How are environments changing in TGE?

6.1 How much is it changing?

The modelling approaches used in this study attempted to predict the 'biotically scaled' stress that the TGE biome will be subject to under different climate scenarios. ANN models, focusing on vegetation structure, suggested very high mean stress (0.75–0.95) in woodlands and grasslands across the biome and within the NRS, under medium and especially high 2070 climate scenarios (Table 2). However, interpretation of these results should be tempered by the moderate stress (0.45–0.62) estimated even for the current climate, suggesting moderate to low success at separating the environments of these structural types.

These results are further emphasised through comparisons of the mapped vs. ANN-modelled areas of TGE environments in the south-eastern Australia study region (Table 3a). Calculations based on pre-European extent indicate that the ANN models predicted considerably less area favouring TGE under current climates than is mapped, again reflecting the low success at separating environments. Nevertheless, the area of woodland (MVG 5) (but not grassland, MVG 19) environments is predicted to decline substantially under 2070 scenarios. Further, the NVIS extant layer suggested only 10.2% of woodland and 24.2 % of grassland remain in the study region today. When this is accounted for, predicted remaining woodland and grassland environments in 2070 drop dramatically to <5% of their pre-European extent for woodlands and <10% for grasslands (Table 3b).

GDM produced estimates of stress based on expected compositional dissimilarity under the present vs. future climate scenarios for each grid cell. These showed similar patterns in grasslands and woodlands, but compositional dissimilarity was typically low to moderate for birds, mammals and frogs, and high for reptiles, snails and plants. Across all groups, compositional dissimilarity increased from 0.14–0.68 under medium 2070 scenarios to 0.29–0.85 under high 2070 scenarios (Figure 3a, Figure 3b). Trends were generally similar for NRS areas of TGE (Figure 3c, Figure 3d).

Table 2 Mean environmental stress (ANN dissimilarity) for current environments of relevant MVSs in south-eastern Australia, for the modelled current, 2070 medium (M) and 2070 high (H) emissions scenarios. Note that moderate stress for the modelled current scenario indicates moderate to poor success at separating the environments of these structural types

ENVIRONMENT	MODELLED MEAN	SE	M 2070 MEAN	SE	H 2070 MEAN	SE
Pre-clearing						
woodland MVS 9	0.46	0.000	0.75	0.000	0.85	0.000
grassland MVS 36	0.62	0.003	0.92	0.001	0.95	0.001
grassland MVS 37	0.62	0.001	0.83	0.001	0.88	0.001
NRS						
woodland MVS 9	0.45	0.010	0.73	0.007	0.79	0.007
grassland MVS 36	0.51	0.049	0.92	0.022	0.95	0.019
grassland MVS 37	0.54	0.025	0.86	0.012	0.88	0.011

Table 3 Area of eucalypt woodland and grassland environments under different scenarios across the south-eastern Australian study region (note this table shows Major Vegetation Groups, a higher level of classification than the subgroups used in this report to define TGE, see Figure 1). (a) Calculations based on estimated pre-European (pre-E) extent as mapped by NVIS (DEWR 2007); (b) Considering extant areas only using NVIS extant layer (DEWR 2007)

(a)

ENVIRONMENT		NVIS MAPPED PRE-E	ANN MODELLED PRE-E	ANN M 2070 PRE-E	ANN H 2070 PRE-E
woodland	km^2 (without clearing)	441 458	290 416	224 743	124 701
(MVG 5)	% of mapped pre-E		65.8	50.9	28.2
	% of modelled pre-E			77.4	42.9
grassland	km^2 (without clearing)	99 628	61 425	119 407	119 410
(MVG 19)	% of mapped pre-E		61.7	119.9	119.9
	% of modelled pre-E			194.4	194.4

(b)

ENVIRONMENT		NVIS MAPPED EXTANT	ANN MODELLED EXTANT	ANN M 2070 EXTANT	ANN H 2070 EXTANT
woodland	km^2 (with clearing)	45 050	34 923	23 392	9 340
(MVG 5)	% of mapped pre-E*	10.2	7.9	5.3	2.1
	% of modelled pre-E*			8.0	3.2
grassland	km^2 (with clearing)	24 065	10 550	9 021	7 329
(MVG 19)	% of mapped pre-E*	24.2	10.5	9.1	7.4
	% of modelled pre-E*			7.6	6.1

*mapped and modelled pre-E from Table 3 (a)

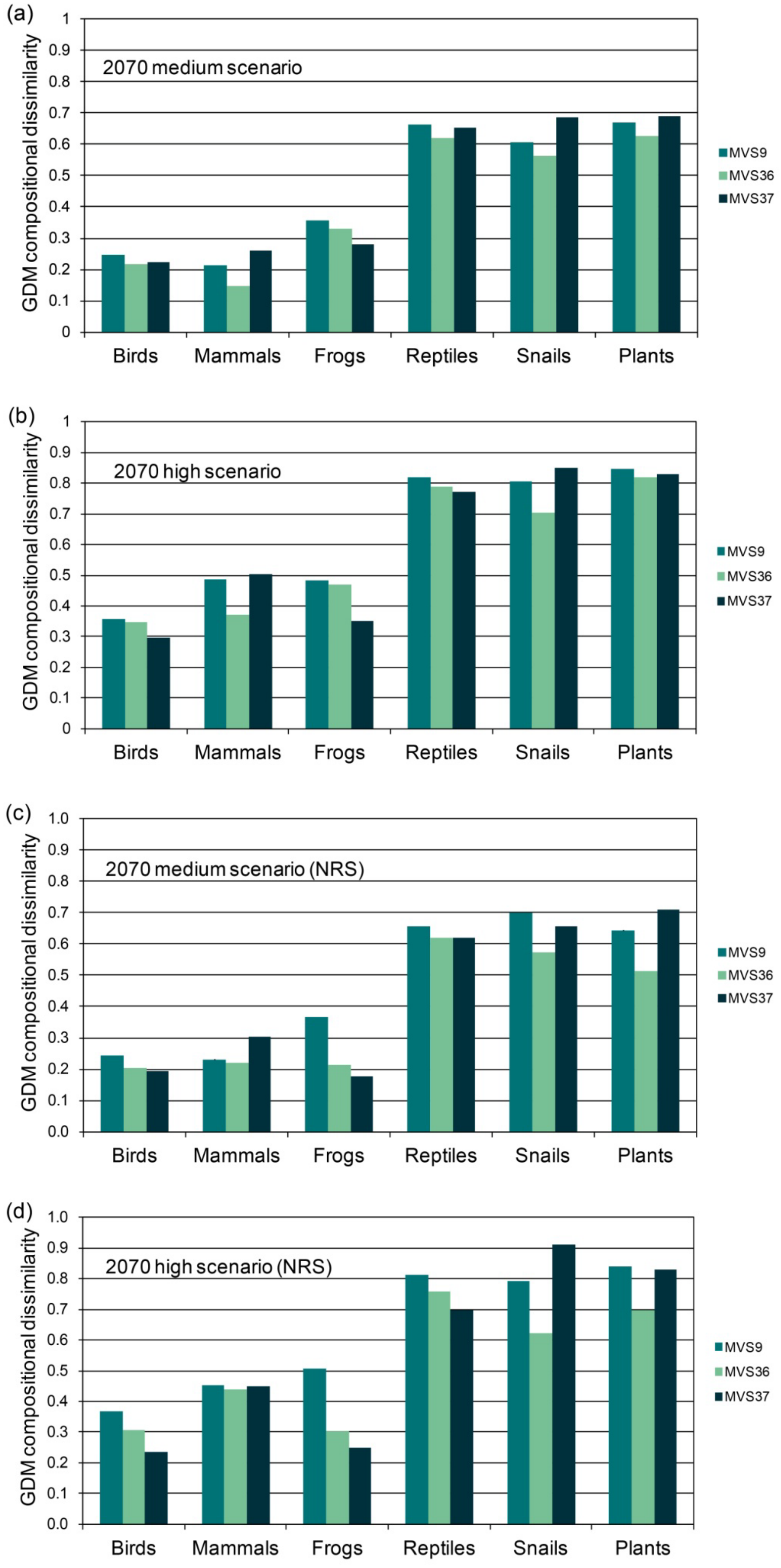


Figure 3 Mean GDM dissimilarity between current and predicted future species composition of each 1 km^2 grid cell within grassy eucalypt woodland (MVS 9) and grasslands (MVS 36, MVS 37), for different biological groups. (a & b) for all areas within the TGE biome, (c & d) within NRS areas in the TGE biome

6.2 Where is it changing?

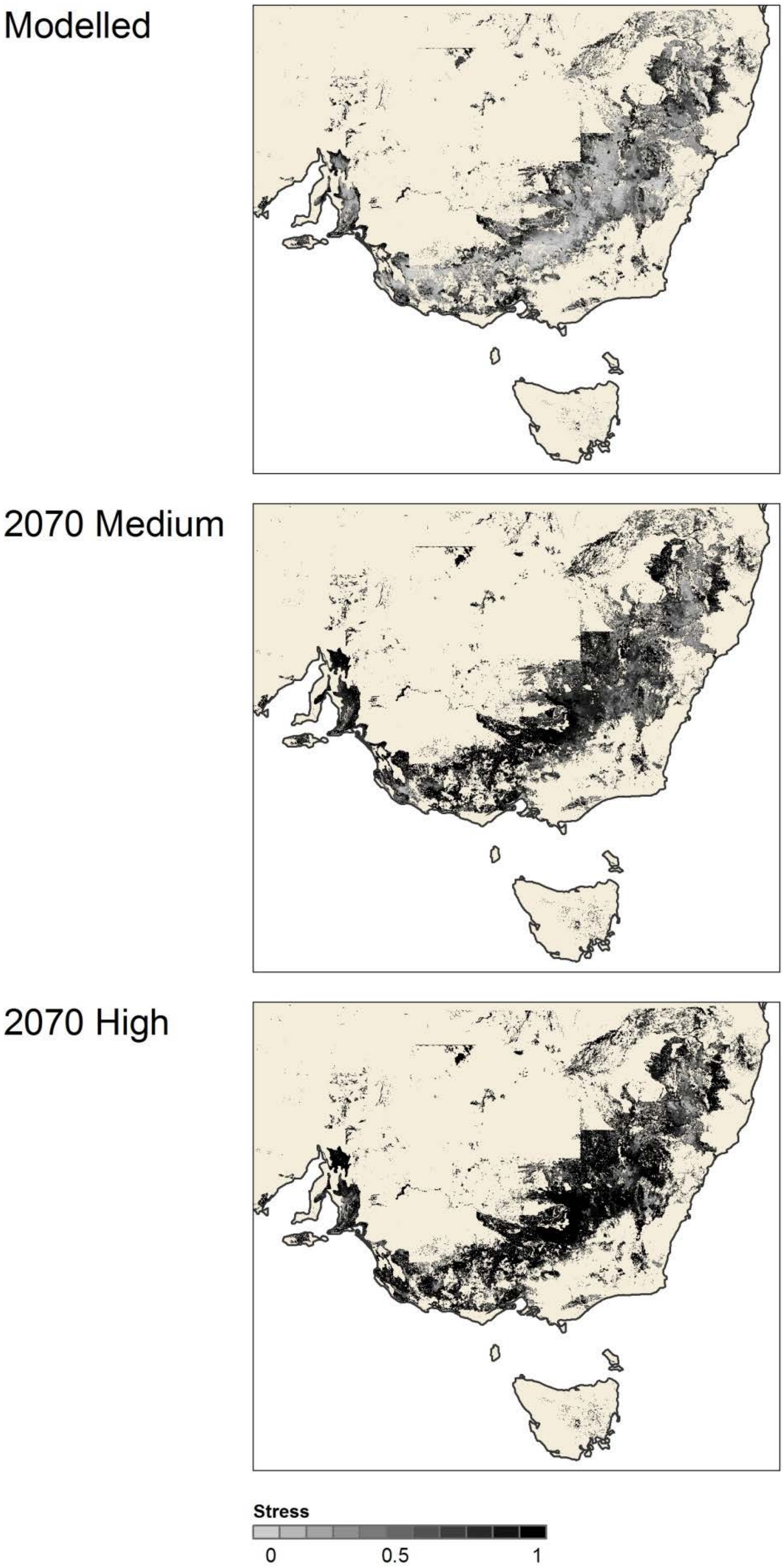
Moderate mean ANN stress levels predicted even under current climates (Table 2) are largely caused by high dissimilarity around the edges of the current TGE biome, with lower dissimilarity in core woodland regions such as central New South Wales and Victoria (Figure 4). Consistent with this, current TGE environments are predicted by ANN to favour other structural types, particularly *Callitris* or *Casuarina* forest or chenopod shrublands, along the inland margins. Woodland and grassland environments are well-predicted in core areas, while more eucalypt forest is predicted along eastern margins, especially on the New England Tablelands (Figure 5).

Although predicted stress, using ANN classification, is generally high under 2070 medium and high scenarios, some regional variation is evident (Figure 4). It is notably more moderate on the New South Wales north-west slopes (but not the Moree Plain or New England Tablelands), and tends to increase southwards and inland. These trends are consistent with predicted distributions of environments favouring woodland and other structural types (Figure 5). In particular, few southern parts of south-eastern Australia are predicted to have environments more suitable for eucalypt woodland than for other structural formations, in either medium or high 2070 scenarios. Under medium scenarios, environments favouring eucalypt woodland persist predominantly on the New South Wales north-west slopes and along narrow margins of the Great Dividing Range, with small expansions in coastal areas such as the Gippsland Plain. These patterns are accentuated under high scenarios, with decreasing woodland environment in northern New South Wales and further small increases on coastal plains.

Where environments no longer favour eucalypt woodlands, environments within the current TGE biome are predicted to become increasingly favourable for chenopod shrublands from inland edges (and even parts of Tasmania, where this structural type does not currently occur), with patchy areas favouring *Casuarina* forests. A number of other structural formations are potentially favoured in more mesic areas, particularly *Callitris* forests inland of the Great Dividing Range, and mallee in South Australia and western Victoria (Figure 5). We emphasise, though, that some of these favoured communities were predicted under current climates within the TGE biome as well. Areas suitable for MVG 21 (‘other grassland/herbland’) or MVG 10 (‘other (non-eucalypt) woodland’) are also predicted; it is unclear what these might represent given these vegetation groups include various miscellaneous ecological communities.

Notably, the extent of environments favouring grasslands is predicted to increase (Table 3a). This is despite very high predicted ANN stress in current grasslands (Table 2), reflecting a changing spatial location of these environments. In particular, temperate tussock grasslands in Victoria tend to become more favourable for other structural types such as shrublands, while new grassland environments appear in northern New South Wales, including much of the New England Tableland (Figure 5).

GDM compositional dissimilarity shows somewhat different spatial patterns in predicted stress levels from ANN, and patterns also differ among different groups of organisms (Figure 6). For nearly all groups, predicted stress is at the lower end of the range in Tasmania. For mammals, reptiles, snails and plants there is a tendency for lower stress in Victoria and higher stress in central New South Wales, and sometimes South Australia. The New South Wales north-west slopes, which appear least stressed in the context of woodland structural formations (ANN), usually show lower stress than central New South Wales but do not stand out ahead of Tasmania or central Victoria. Birds and frogs show a somewhat different pattern, with stress tending to increase from the inland edges of TGE towards the more coastal edges.

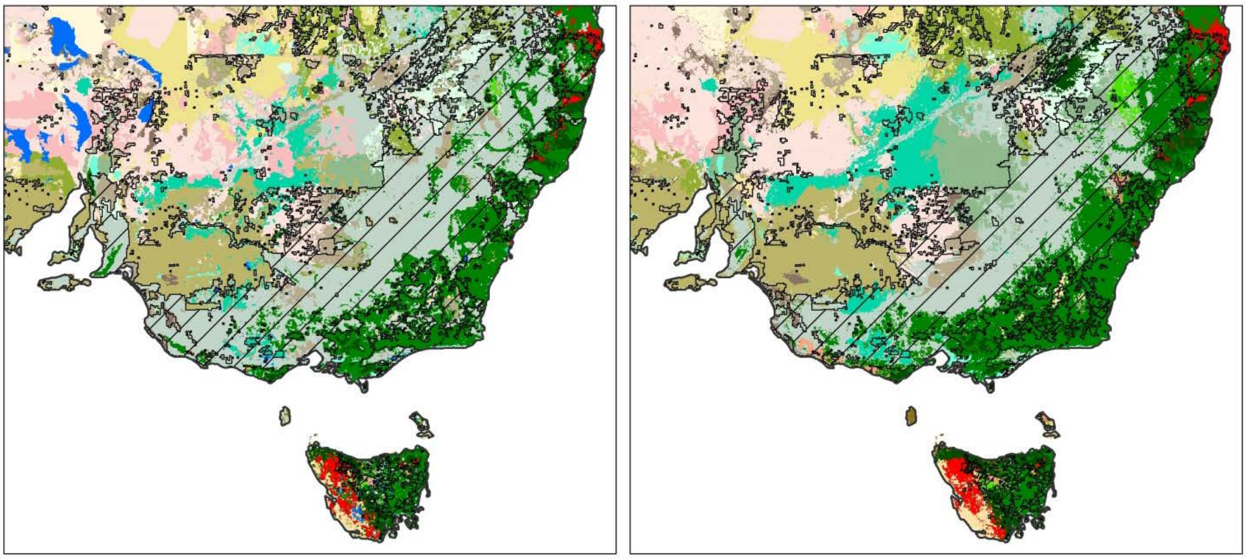


Modelled

2070 Medium

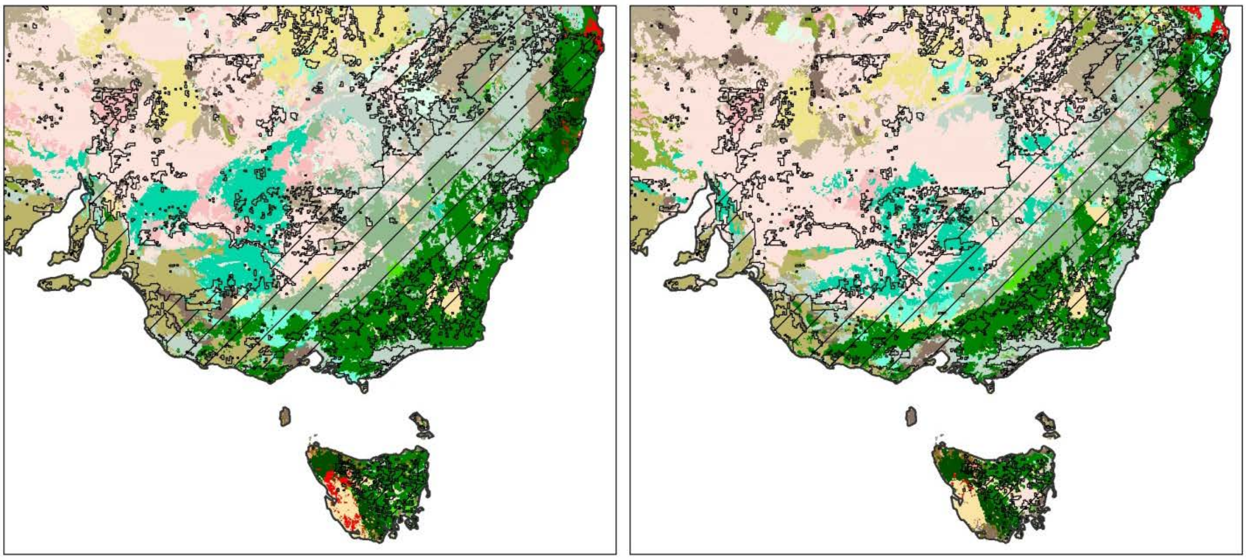
2070 High

Figure 4 ANN dissimilarity (environmental stress) based on the difference between current, modelled and future climate change scenarios in the TGE biome. Dark shading represents the greatest predicted environmental stress



(a) NVIS pre-clearing vegetation map

(b) ANN modelled environments for current climate



(c) ANN modelled environments for 2070 medium

(d) ANN modelled environments for 2070 high

🟥 Rainforest and vine thickets
🟩 Eucalyptus tall open forest
🟩 Eucalyptus open forest
🟩 Eucalyptus low open forest
■ Eucalyptus woodlands
■ Acacia forests and woodlands
■ Callitris forests and woodlands
■ Casuarina forests and woodlands
■ Melaleuca forests and woodlands
■ Other forests and woodlands
■ Eucalyptus open woodlands
■ Tropical Eucalyptus woodlands/grasslands
■ Acacia open woodlands
■ Mallee woodlands and shrublands
■ Low closed forest and tall closed shrubland
■ Acacia shrublands
■ Other shrublands
■ Heath
■ Tussock grasslands
■ Hummock grasslands
■ Other grasslands, herblands, sedgelands and rushlands
■ Chenopod shrublands, samphire shrubs and forblands
■ Mangroves
🟦 Inland aquatic - fresh water, salt lakes, lagoons

Figure 5 The distribution of environments suitable for major vegetation groups (MVGs) in south-eastern Australia under different climate scenarios. Hatched area indicates approximate current distribution of TGE

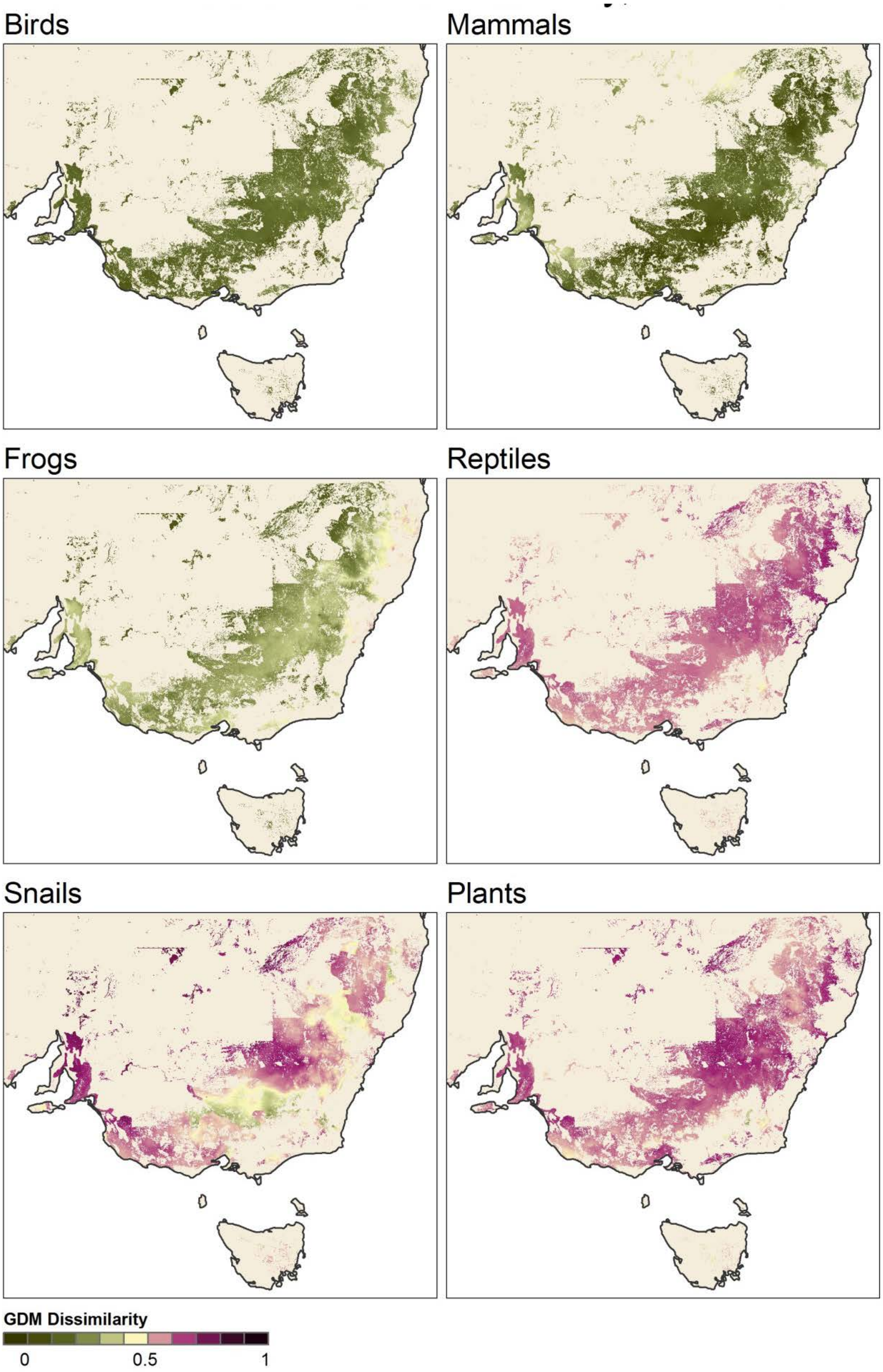


Figure 6a 2070 medium scenario: Predicted compositional change (stress) for each 1 km^2 cell calculated from GDM modelling for different groups of organisms. High dissimilarity (dark pink) indicates a high level of stress

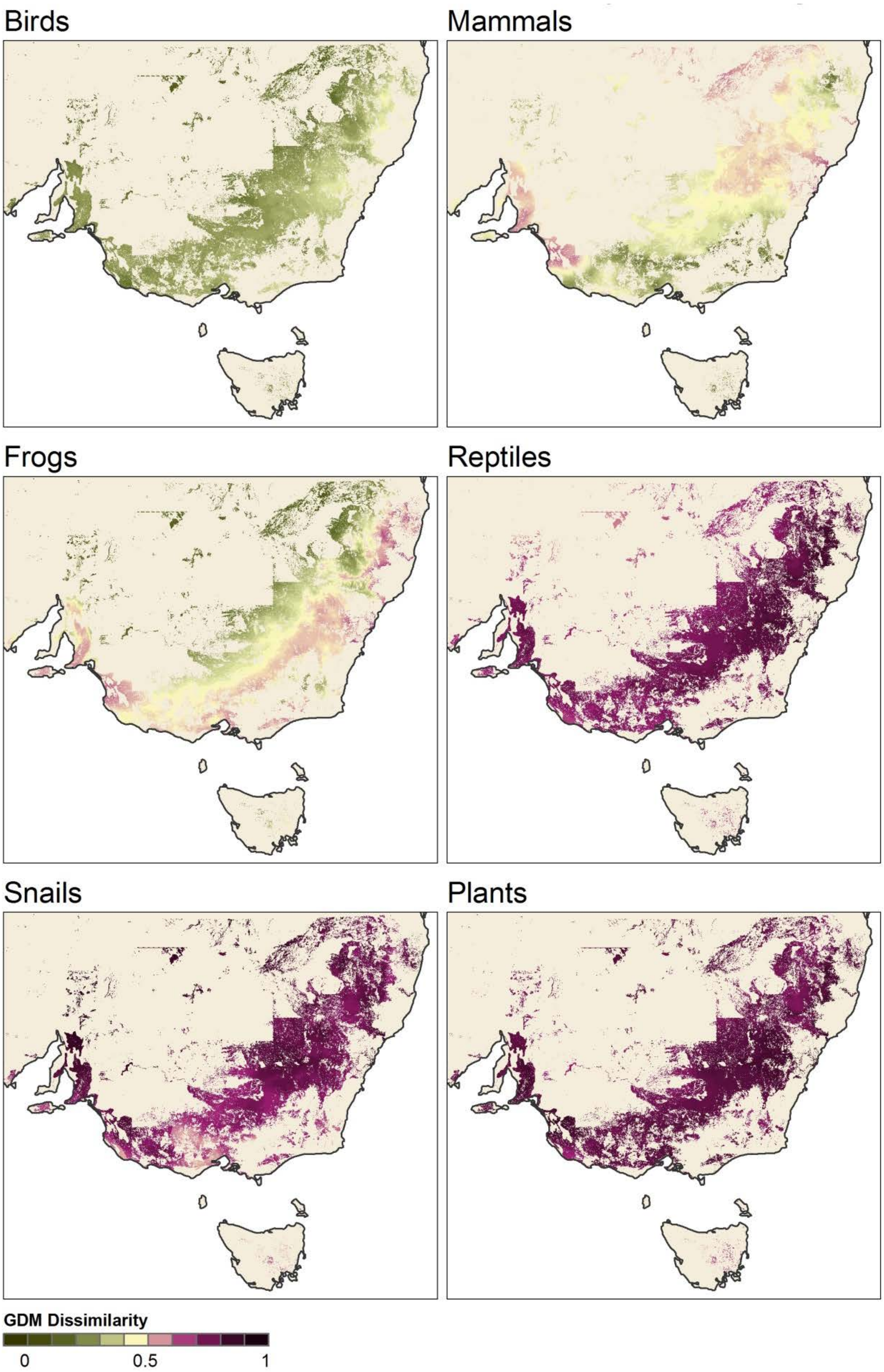


Figure 6b 2070 high scenario: Predicted compositional change (stress) for each 1 km^2 cell calculated from GDM modelling for different groups of organisms. High dissimilarity (dark pink) indicates a high level of stress

6.3 Which climate variables are most driving the TGE distribution?

The three variables that contributed most strongly to describing the distribution of eucalypt woodlands (MVG 5) and tussock grasslands (MVG 19) in ANN models nearly all related to moisture. For woodlands, moisture index (an estimate of soil moisture based on rainfall and evaporation) of the cool season and of the wettest period were foremost, followed by rainfall seasonality. For grasslands they were similar, including in order of importance, rainfall seasonality, moisture index of the wettest period and A horizon saturated hydraulic conductivity. The latter provides an indication of soil texture and probably reflects occurrence of many (although not all) lowland grasslands on clay soils.

7 To what extent might impacts of climate change on biodiversity be buffered by attributes of resilience at continental, regional and local scales?

Although models suggest high environmental stress within the TGE biome under climate change, outcomes for TGE will be mediated by a range of attributes of component individuals, populations, species, ecosystems and landscapes that confer resilience to changing environments. As well, opportunities outside the current range of TGE (as described above) may facilitate the persistence of TGE species. An understanding of these attributes can inform development of management options.

Prior to degradation, at least, a range of TGE characteristics offered mechanisms for effective resistance or adaptation to environmental change. Many TGE plant and vertebrate species were naturally widespread and abundant across broad climatic gradients, with large population sizes and high genetic diversity (e.g. Prober and Brown 1994; Prober and Thiele 2005). These species may thus have greater capacity to persist in a warming and/or drying climate, especially at the cooler, moister ends of their ranges or in run-on zones in lower parts of the landscape. This is consistent with GDM measures of stress or point-scale dissimilarity that suggest less compositional change in cooler regions such as Tasmania and elevated parts of central-west Victoria for some groups of organisms; that is, these maps suggest species in these cooler areas might cope with warmer (or otherwise different) conditions than they currently experience at these locations (Figure 6). However, there is not strong evidence for greater compositional similarity in moister (more coastward) zones as might have been expected for species currently distributed widely across coast–inland rainfall gradients. Indeed, somewhat contrary trends are observed for birds, frogs and reptiles, suggesting the inland fauna may be more resilient.

Natural patterns of environmental variation associated with climatic gradients and topographic heterogeneity (Prober and Thiele 2005) are another feature of TGE that may facilitate adaptation. In heterogeneous environments it is more likely that new areas of suitable habitat will develop close by (e.g. upslope or coastward), that can then be accessed by organisms through local or regional migrations. This is moderately supported by GDM estimates of change in 'effective habitat area' for plants at differing scales. These suggest losses of environment suitable for current species composition would be decreasingly severe as radii for potential migrations increase from 750 m to 100 km, especially for temperate tussock grasslands (Figure 7). Spatially, patterns of change in effective habitat area still remain strongly influenced by point-scale stress, with highest levels in central New South Wales and least in central Victoria, Tasmania and northern New South Wales (Figure 8). Areas that offer the most buffering independent of point-scale stress are emphasised in Figure 9. As might be expected, buffering though this mechanism is most apparent in zones with greater relief, particularly the fringes of the Great Dividing Range in New South Wales and Victoria, and in Tasmania. Note though that these patterns assume effective mobility and connectivity at the relevant scales.

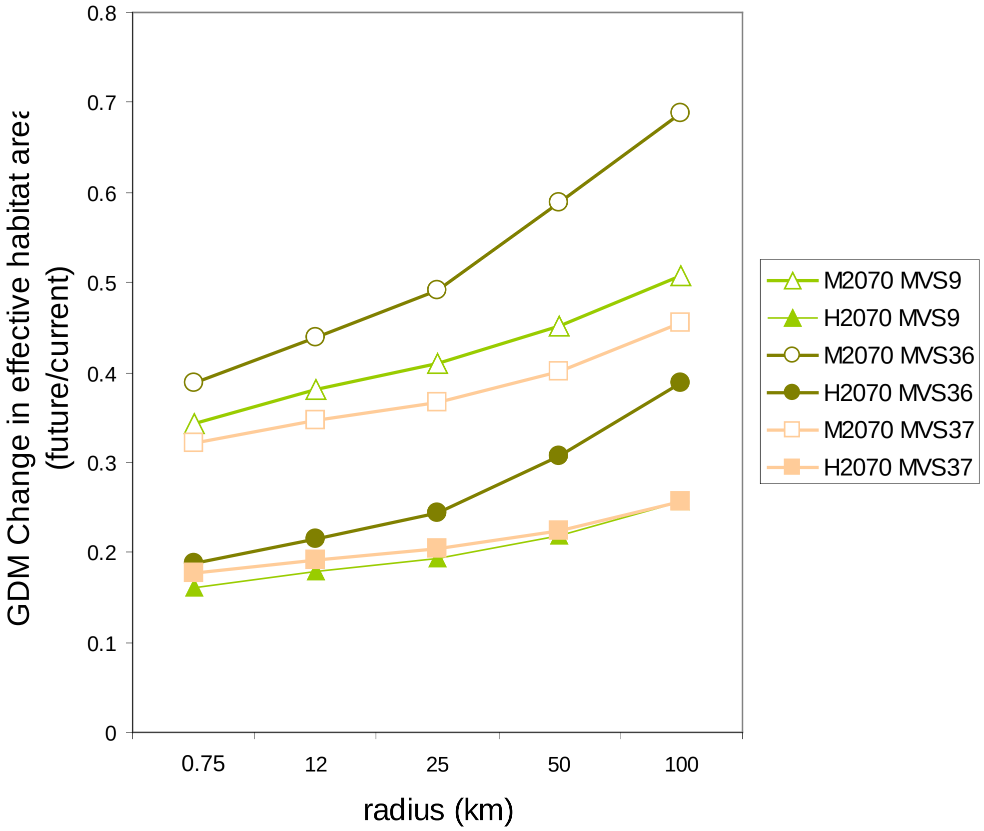
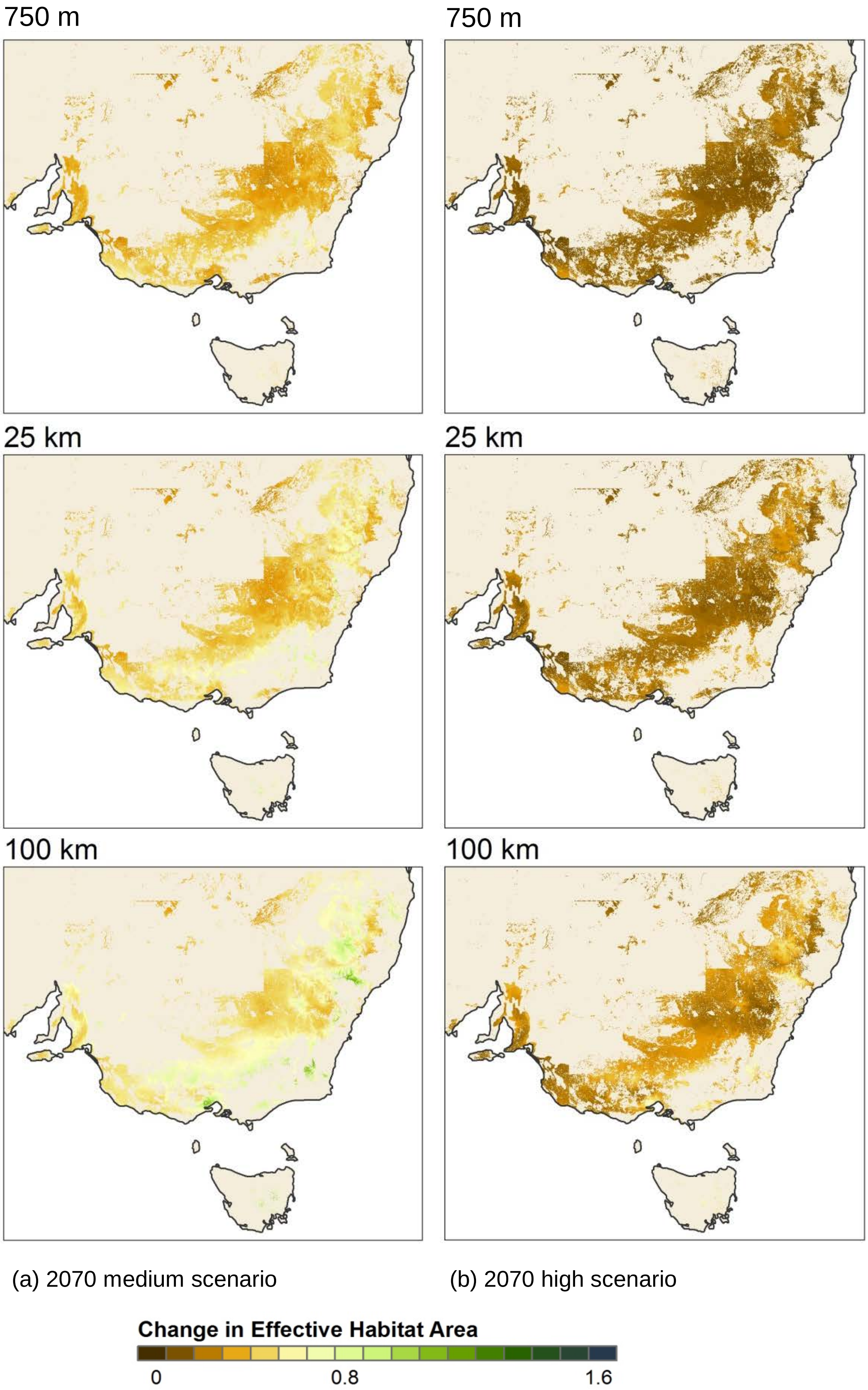


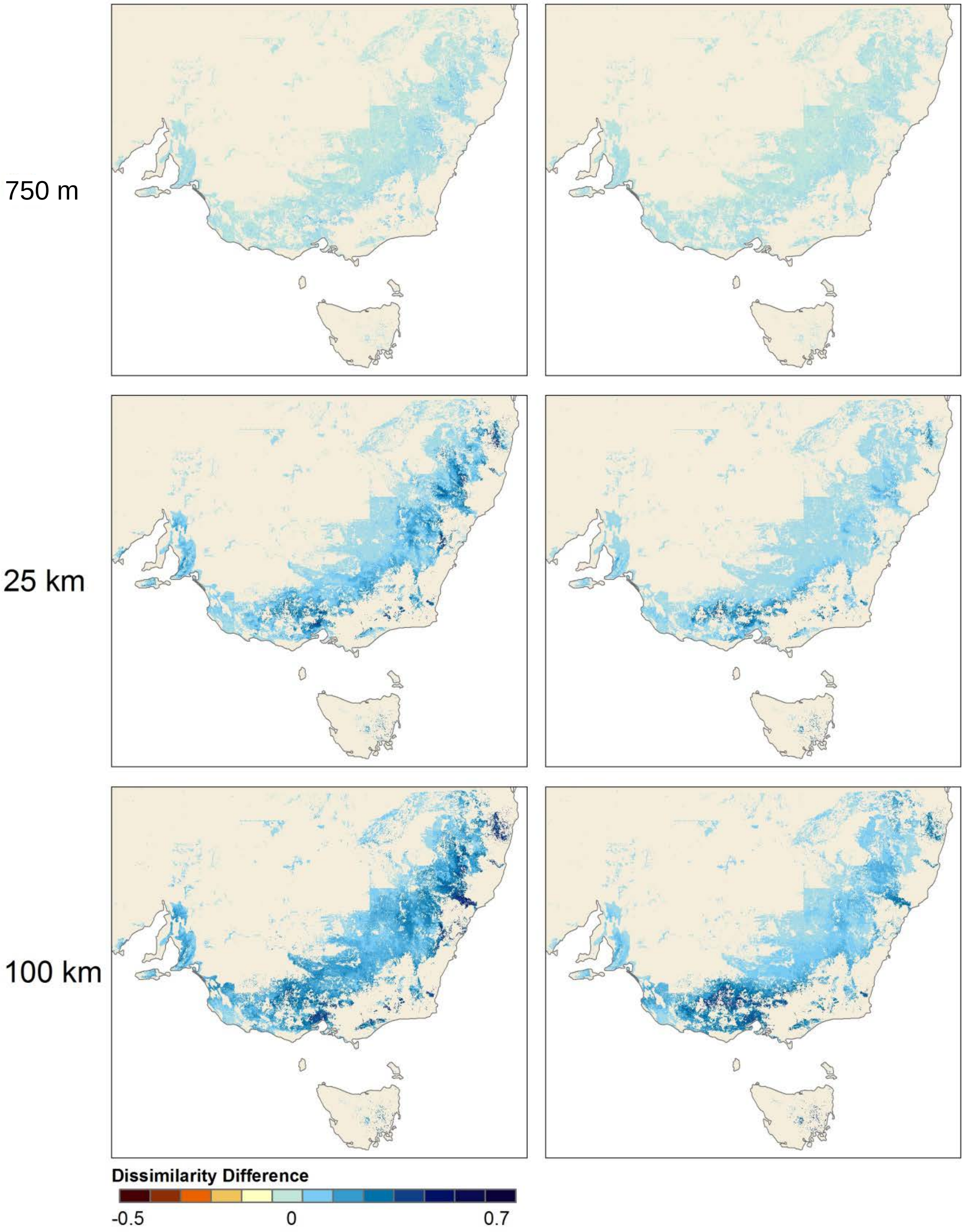
Figure 7 Mean change in effective habitat area (future/current) for the TGE biome, calculated on the basis of all cells within radii from 750 m to 100 km. This indicates combined effects of point-scale climate change and levels of buffering offered by local landscapes, on future habitat area. Note lower values indicate lower future habitat area compared with current



(a) 2070 medium scenario

(b) 2070 high scenario

Figure 8 Spatial distribution of predicted change in effective habitat area (future/current) in the TGE biome, calculated on the basis of all cells within radii from 750 m to 100 km. Darker browns indicate greater loss of habitat area within the defined radius (i.e. lower future area compared with present), dark greens indicate increased habitat area



(a) 2070 medium

(b) 2070 high

Figure 9 Predicted ‘buffering’ effect due to environmental heterogeneity (see methods), calculated on the basis of all cells within radii from 750 m to 100 km. Darker blues indicate greater potential buffering for the defined radius, browns indicate higher similarity than at present

At patch scales, the natural plant diversity of TGE can be among the richest in the world (Lunt 1990; Kirkpatrick et al. 1995). This offers further resilience to change, by increasing the likelihood that currently redundant, subsidiary species can step in to perform ecosystem functions should functionally significant species become extinct (Walker et al. 1999). On the other hand, some natural attributes of TGE suggest greater vulnerability to change. Perennial forb species contribute much of the native plant diversity, yet few of these have long-lived seed banks and many are poor dispersers (e.g. Lunt 1997). This decreases their capacity to recover after drought or migrate to more suitable environments. Many characteristic woodland fauna are also vulnerable, owing to their dependence on biotic interactions or phenological events, for example, temporal continuity in availability of nectar across the biome (Lunt and Bennett 2000).

Finally, ANN models suggest only minimal areas of additional eucalypt woodland environment developing within south-eastern Australia (Figure 5), limiting the potential for temperate eucalypt woodland to develop elsewhere. However, environments favouring grasslands may increase, potentially providing greater opportunity for some grassland species.

GDM estimates of the absolute minimum dissimilarity between each grid cell and any other within Australia provide an analogous indication of whether species composition similar to TGE could persist somewhere in Australia. These statistics suggest that under 2070 medium scenarios, areas with moderate to high similarity to TGE in parts of central Victoria, Tasmania and northern New South Wales may persist somewhere within Australia. However, it is not clear whether these suitable environments would be reachable by component biota. Minimum dissimilarity in other areas is typically moderate to high, and under 2070 high scenarios few even moderately similar environments are predicted except for Tasmanian TGE (Figure 10).

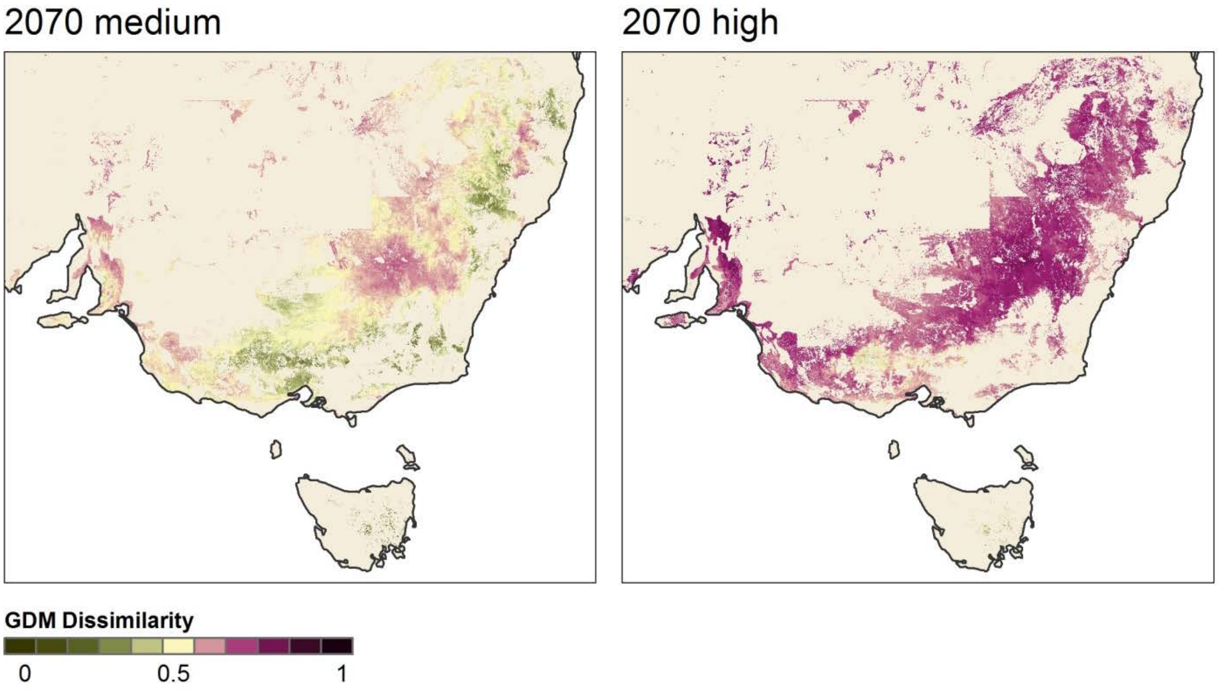


Figure 10 GDM absolute minimum dissimilarity (see methods) indicating potential for the species composition of a cell to occur at any point within Australia under future scenarios

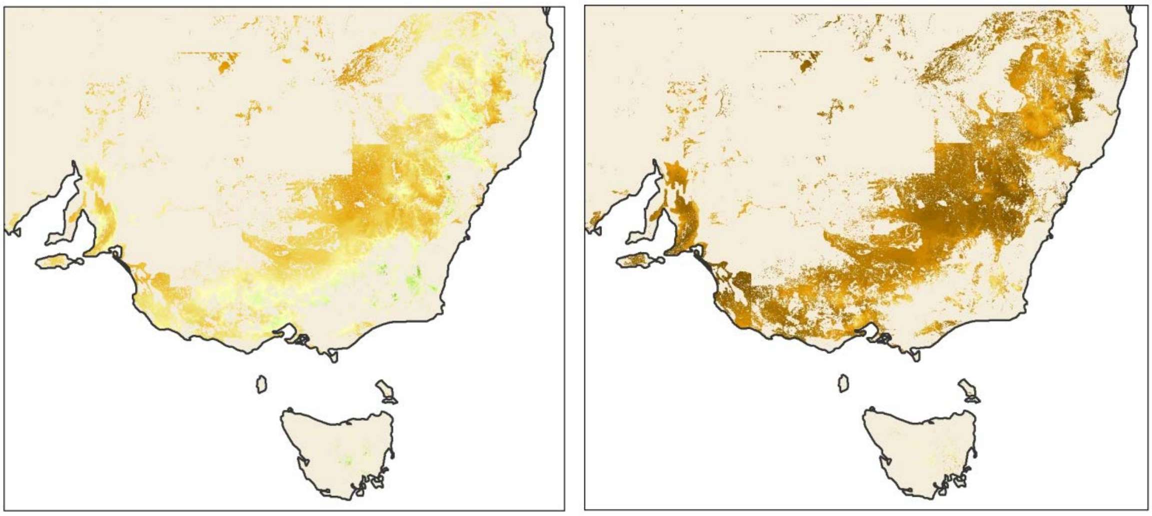
8 Interactions between climate change and land use

The high degree of fragmentation and degradation in TGE will have dramatically altered their capacity to adapt to climate change. Not only has the total area of potentially suitable environment been drastically reduced (Table 3a), many of the above natural attributes of resistance and resilience have been compromised. Smaller population sizes are associated with lower genetic diversity (e.g. Prober and Brown 1994), and could limit persistence and adaptation at the species level. Loss of native species diversity at site scales has reduced the pool of species available to perform ecosystem functions, and reduced landscape connectivity will severely restrict opportunities for organisms to migrate either locally or regionally under climate change. Further, degradation of site-scale ecosystem processes such as soil water infiltration or habitat structure will affect the capacity of ecosystems to cope with additional climate-related stresses, and interactions with exotics will facilitate community reassembly by exotic rather than native species.

The impacts of degradation become glaringly evident when comparing projections for change in effective habitat area (future/current area within a defined radius) based on the mapped historical vs. extant distribution of TGE (Figure 11). For example, estimates of future vs. current habitat area using a 50 km radius decrease from a mean of 0.48 for pre-clearing TGE to a mean of 0.2 for extant TGE under 2070 medium scenarios. Notably, few potential 'refuge' areas remain robust. These unfortunately dire projections are likely to more closely reflect the true status of the TGE biome than other predictions, which better indicate 'potential' resilience of intact ecosystems. We note, though, that estimates of extant vegetation do not account for significant areas of native pasture and smaller remnant patches that can contribute to adaptation in TGE landscapes.

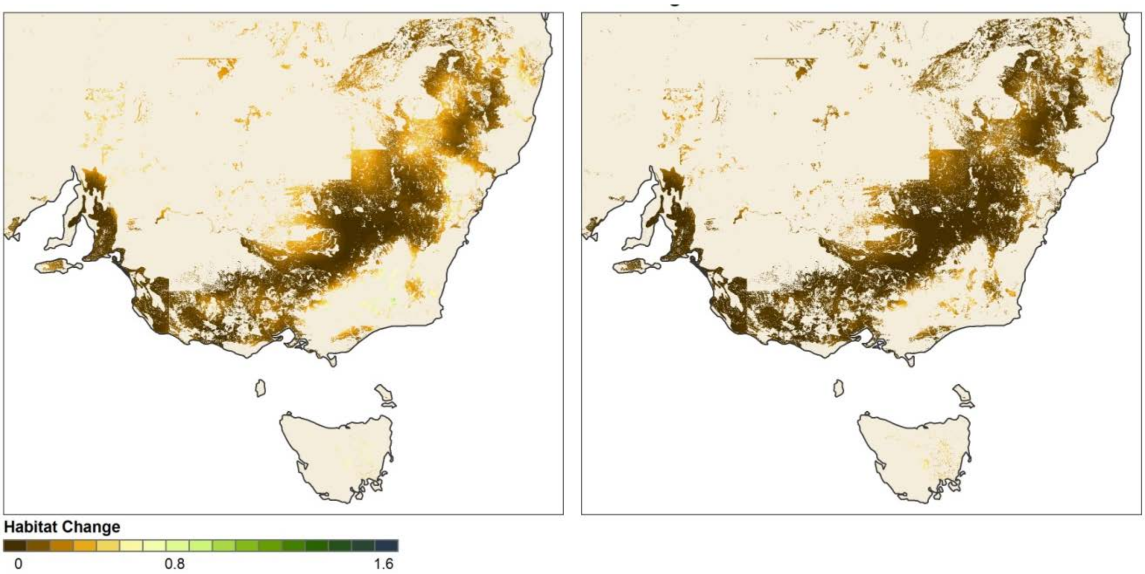
Although most impacts of degradation are likely to reduce the capacity of TGE to adapt to change, some could facilitate it. In particular, livestock grazing has already resulted in broadscale increases in native species that were once more prominent in drier environments (e.g. Moore 1970; Prober and Thiele 2004), including native grasses and annual forbs. It is thus possible these species are pre-adapted to withstand some climatic drying in more mesic TGE. These changes occurred earlier during the history of post-European development, probably in the context of somewhat less fragmented environments. Nonetheless, they suggest an analogy to how climate adaptation might occur over relatively short time frames. In particular, they suggest dramatic shifts in relative abundances and local distributions (e.g. Moore 1970), as well as significant ecosystem simplification (e.g. reduced diversity and loss of the intermittent shrub layer) at greater extremes of disturbance (Prober and Thiele 2005).

Human-mediated outcomes of climate change in TGE are not restricted to effects of past activities. Current and future land use changes – driven by changing viability of grazing and cropping enterprises, urbanisation, and emergence of new markets for carbon and biofuels – could ameliorate or magnify climate change impacts.



(a) 2070 medium, without clearing

(b) 2070 high, without clearing



(c) 2070 medium, with clearing

(d) 2070 high, with clearing

Figure 11 Spatial distribution of predicted change in effective habitat area within a 50 km radius (future suitable area/current suitable area), contrasting predictions for hypothetically uncleared landscapes vs. cleared landscapes

9 Will there be novel environments?

While the previous GDM results showed how the present might compare with the future, Figure 12 shows how the future under 2070 high scenarios might compare with the present, indicating where novel environments are likely to develop. Across most of TGE, a high dissimilarity between future environments and any currently existing on the Australian continent is suggested, with some part-parallel environments in central-west Victoria and Tasmania (Figure 12). Thus, not only are there few environments across Australia expected to be similar to TGE today, but the environments that develop in current TGE are likely to promote assembly of new ecological communities. As these are novel environments, relationships between biodiversity and this habitat are not easily predicted. In this context it should be remembered that ANN modelling focused on environmental suitability for pre-classified vegetation types, and was not used to predict environments more suitable for novel systems.

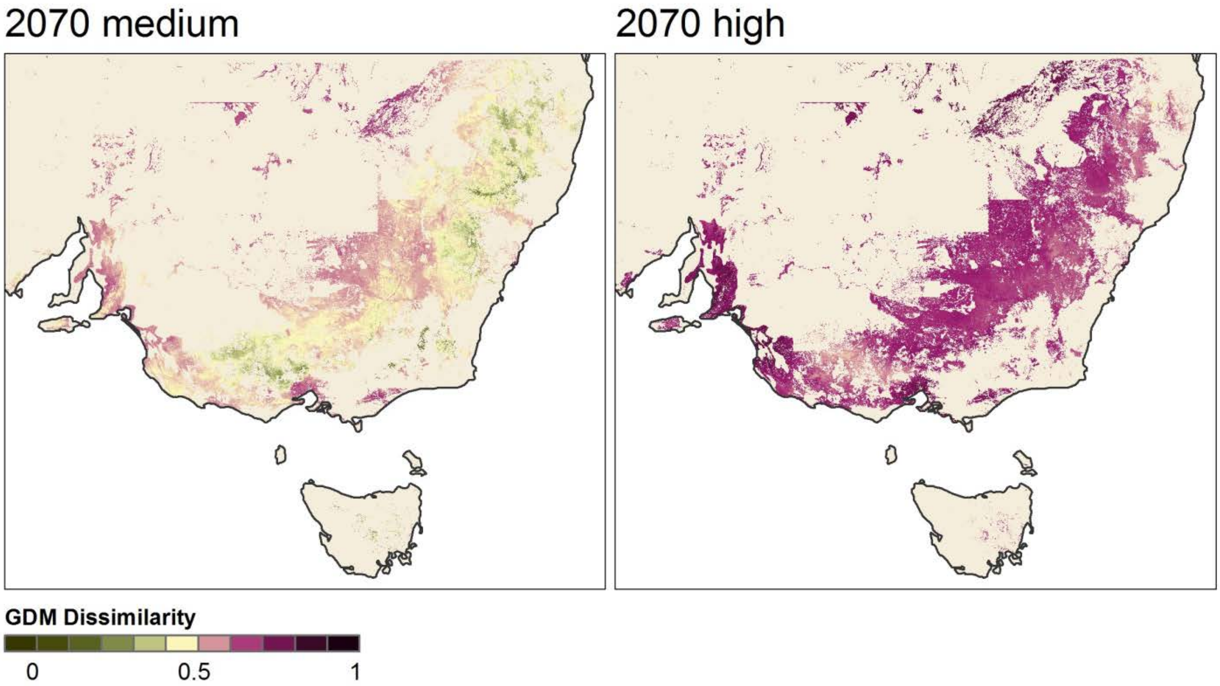


Figure 12 Minimum dissimilarity between future composition of a cell and the current composition of any cell on the continent, as an indicator of novel environments. High dissimilarity (dark pink) indicates that the future environment currently has no similar analogues on the continent; dark green indicates future environments currently occur somewhere in Australia

10 What major ecological changes are likely?

Integration of our understanding of driving ecological processes in TGE with ANN and GDM predictions facilitates speculation with regard to likely directions of ecological change under future climate scenarios. The following discussion puts forward five potential outcomes of climate change in TGE, acknowledging that they are subject to substantial uncertainty. Suggested outcomes include altered fire regimes, structural change, shifts in the functional composition of the herbaceous ground-layer, predominance of exotic species in reassembled communities, and complex changes in ecological interactions.

10.1 Changing fire frequency

Fire is a driving process in TGE that could both respond to and cause ecosystem change in a changing climate. Increasing temperatures are expected to result in increasing frequency of severe fire weather events across south-eastern Australia, but in TGE this effect is likely to be offset by reduced growth of herbaceous fuels associated with lower cool-season rainfall, higher summer moisture stress and/or other structural change (see below, Bradstock 2010). The net outcome for natural fire regimes in TGE is thus expected to be a decline in fire frequency if plant growth is restricted (Bradstock 2010). However, fire could increase due to reduced fuel moisture content, especially in cooler sub-alpine regions (Lunt et al. 2012) or where factors such as increasing summer rainfall, winter temperatures or CO_2 result in enhanced grassland productivity (Hovenden and Williams 2010).

Projected outcomes for intact landscapes will also be modified by human impacts. In particular, high levels of fragmentation and active fire suppression are likely to limit the influence of climate change on fire regimes (Lunt et al. 2012), whereas widespread invasion by exotic grasses such as buffel grass (*Pennisetum ciliare*) could restore the capacity for fire ignition and spread (Martin et al. 2012).

10.2 Structural change

ANN models predict that high environmental stress within the current bounds of the TGE biome is likely to result in structural change in the long term. Consistent with this, consideration of dominant ecological processes in TGE suggests structural changes could include declining tree cover, or woody thickening through effects on processes that moderate the shrub/grass balance.

10.2.1 DECLINE IN TREE COVER

Rainfall is a key determinant of tree cover in savannah ecosystems (Sankaran et al. 2005), so the capacity of trees to persist in the TGE biome is likely to be reduced if moisture stress increases. Consistent with this, ANN models suggest environments becoming more suited to treeless communities, including grasslands, in limited parts of northern New South Wales (particularly on the clay soils of the New England Tableland), and chenopod shrublands in drier zones. Chenopods are already common in native understoreys of drier eucalypt woodlands, and chenopod shrublands are known to predominate in areas where limited, sporadic rainfall and low soil moisture are unable to support tree-dominated vegetation, so such a transition is not infeasible (Keith 2002).

Furthermore, studies in Tasmanian woodlands have indicated increasing levels of eucalypt dieback along gradients of decreasing mean annual rainfall (Calder and Kirkpatrick 2008). If climate change increases moisture stress, this could exacerbate the widespread dieback of eucalypts in many agricultural landscapes. It is possible that rising CO_2 could offset predicted tree decline by improving water use efficiency (Hovenden and Williams 2010), but widespread loss of woodland trees, if it occurs, would have serious implications for woodland-dependent fauna, soil processes and the identity of these iconic Australian landscapes.

10.2.2 ALTERED SHRUB-GRASS BALANCE

Many grassy ecosystems are thought to be maintained by disturbance, which reduces woody plant cover below its climatic potential (Bond and Keeley 2005; Sankaran et al. 2005). The balance between woody and grassy components of TGE is thus expected to be relatively sensitive to disturbance regime and its climatic drivers. Studies of shrub encroachment in semi-arid woodlands of western New South Wales support this, showing strong links between widespread, pulsed recruitment of shrubs and heavy livestock grazing, depleted grass cover, and reduced landscape fire (Lunt et al. 2012).

Under climate change, a number of expected environmental changes could similarly promote woody thickening in TGE. First, there is some evidence that increasing CO_2 will favour relative increases in Australian woody plant species compared with grasses (Hovenden and Williams 2010). Second, should grass biomass accumulation rates decline with increasing moisture stress, this is likely to result in more bare ground into which shrubs can invade. Third, slower rates of fuel accumulation would also reduce fire frequency (see above), reducing mortality events in fire-sensitive shrubs and trees (e.g. see Bond and Midgley 2000).

Woody thickening is consistent with ANN predictions of increasing environments favouring fire-sensitive *Callitris* and *Casuarina* forests under climate change. However, ANN models potentially accounted for only one of the above drivers of woody thickening (increased bare ground). Further, the ANN classification predicted many current TGE environments to be more suitable for *Callitris, Casuarina* or eucalypt forests even under current climates. It is not possible to distinguish sources of uncertainty in these classifications, but these deviations could be partly attributed to historical fire responses, emphasising that future projections are likely to be at least partly fire-dependent and should be considered with caution.

Woody thickening can be seen in both a positive and a negative light. From the perspective of current woodland biodiversity, increasing shrub cover is likely to lead to reduced forb cover and diversity in otherwise diverse remnants (e.g. Price and Morgan 2008), and to threaten fauna dependent on open habitats, for example seed-eaters. Homogenisation across the landscape is likely, with the grassy parts of the landscape becoming more similar to shrubby areas. On the other hand, shrubs would increase structural diversity at site scales, favouring other fauna and potentially storing more carbon (noting that this has not been demonstrated). Shrubs may also help to prevent reassembly by exotic species and facilitate optimal ongoing ecological functioning in a new environment.

10.3 Changes in functional types in the herbaceous ground layer

Irrespective of changes to the woody component, ecosystems developing within the woodland biome are likely to include some herbaceous species. Given that 65–85% of plant species are predicted to change under 2070 medium and high scenarios (Figure 3), this is likely to include shifts in functional composition of the herbaceous layer. Possible functional changes include an altered C_3–C_4 balance and an increasing number of annual compared with perennial species.

10.3.1 ALTERED C_3–C_4 BALANCE

Enhanced plant growth due to CO_2 enrichment is generally greater for C_3 than C_4 species, especially at higher temperatures but to a lesser extent at low nutrient levels (Poorter and Navas 2003; Williams et al. 2007a). Reduced fire (as expected under increasing aridity) can also favour C_3 grasses compared with C_4 grasses (e.g. Prober et al. 2007). Together these suggest a C_3 advantage in TGE under climate change, but this was not supported by CO_2 enrichment and warming experiments in Tasmanian grasslands (TasFACE, Williams et al. 2007a). These treatments resulted in a net increase in the C_4 grass *Themeda triandra* at the expense of C_3 *Austrodanthonia*. Under CO_2 enrichment alone *Austrodanthonia* flowered less, and with both warming and CO_2 enrichment, germination and seedling survivorship also declined. The latter occurred in correspondence with declining soil moisture.

Other changes might favour C_3 to C_4 shifts. The TasFACE experiments suggested CO_2 enrichment was associated with progressive nitrogen limitation (Hovenden et al. 2008), which could play a role in favouring C_4 over C_3 species (e.g. Prober and Lunt 2009). However, nitrogen limitation was not observed when CO_2 enrichment was combined with 2 °C warming (Hovenden et al. 2008). Increases in summer rainfall could also favour warm season C_4 species over cool season C_3 species.

Given these uncertainties regarding outcomes for the C_3–C_4 balance, it is difficult to speculate on wider biodiversity consequences; however, some impacts on exotic invasions are considered in section 10.4.

10.3.2 DECREASING PERENNIALS AND INCREASING ANNUALS

Native perennial forbs form a characteristic element of TGE plant diversity. These include many oligotrophic species that have suffered dramatic declines owing to fragmentation, grazing, nutrient enrichment and weed invasion (Prober and Thiele 1995; McIntyre and Lavorel 2007; Dorrough and Scroggie 2008). Increasing moisture stress is likely to further threaten these species, given that they decline naturally in abundance and diversity along gradients of declining rainfall (Prober and Thiele 2004). As well, many are poor dispersers and will have difficulty migrating to other suitable (low nutrient) sites in a fragmented landscape.

Native annuals could benefit from this shift, as these naturally replace perennial forbs in drier environments (Prober and Thiele 2004). However, TasFACE results suggest (exotic) annuals are susceptible to warming, causing reduced seed production and population decline in some species (Hovenden et al. 2007; Williams et al. 2007a). Likely interactions between changing temperature and rainfall thus make it difficult to predict net outcomes of climate change for cool season annuals.

10.4 Exotic invasions

Release of resources through disturbance of extant ecosystems is considered a key cause of exotic invasion. By increasing mortality in extant ecosystems, climate change will similarly open up more opportunities for exotic invasions (Scott et al. 2008). For isolated, often nutrient-enriched remnant vegetation in a highly disturbed matrix, this suggests a high likelihood that newly developing ecosystems will include a high exotic component. This could include both exotics currently in the biome and new invaders to the region.

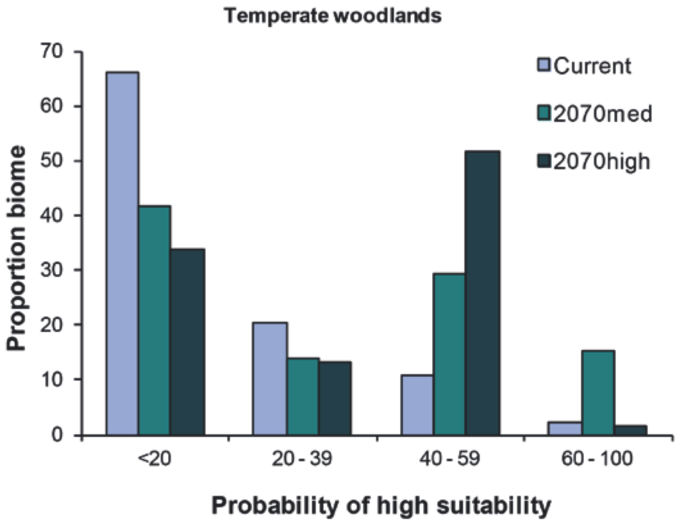


Figure 13 Proportion of the TGE biome predicted to be highly suitable to buffel grass colonisation under current (light grey), 2070 medium (medium grey) and 2070 high (black) climate scenarios (Martin et al. 2012)

Currently, exotic annuals are the most widespread and pervasive weeds in TGE. These could be favoured by increasing CO_2, since this has been shown to favour fast-growing C_3 species (Poorter and Navas 2003). On the other hand, they could be disadvantaged by considerable reductions in winter and spring rainfall (e.g. Prober et al. 2009), and potential CO_2-induced nitrogen limitation (Hovenden et al. 2008; Prober and Lunt 2009). A net disadvantage to exotic annuals is supported by the TasFACE experiments, which have shown reduced survivorship and seed production under CO_2 enrichment and warming (Williams et al. 2007a; Hovenden et al. 2007).

A currently increasing threat in TGE is invasion by exotic perennial grasses, e.g Love Grass (*Eragrostis curvula*) and Coolatai grass (*Hyparrhenia hirta*). These C_4 species could be favoured under future climate scenarios that favour warm season C_4 species (e.g. increasing summer rain), but observations of the distribution of these particular weeds suggest they may be a decreasing threat if rainfall decreases below an annual mean of 400 mm (S. Prober pers. obs.). While some exotics could be disadvantaged, other exotics more tolerant of moisture stress could become more problematic in TGE. In particular, Bayesian belief networks (Martin et al. 2012) suggest increasing probability of high suitability for buffel Grass (*Pennisetum ciliare*) – especially in South Australia, northern Victoria and northern New South Wales – under medium and high 2070 climate scenarios. This species is already known to have a negative impact on native biodiversity within other parts of Australia, including tropical forests and woodlands of Queensland and in the arid landscapes of central Australia (see Martin et al. 2012 and references therein).

10.5 Changing ecological interactions and species composition

Cascading changes in ecological interactions will undoubtedly lead to ecological impacts that are beyond the predictive capacity of either modelling approach employed in this study. These are likely to include altered invertebrate dynamics, influencing pollination, predation, nutrient turnover, herbivory and other functions; decoupling of mutualisms due to altered phenologies or different migration rates of interacting species (e.g. plant–micorrhyzal or plant–animal dependencies); and changing patterns of disease outbreaks. Some naturalists at the workshop argued these types of change are already occurring, noting a new 'quietness of the bush'.

In this respect it is notable that changes in composition of birds, mammals and frogs predicted by GDM models appear consistently less dramatic than for vegetation. Lower estimates for the former possibly reflect the wider distributions of species in these groups; however, models account only for physical environments, and not for changing vegetation structure, biological resources and ecological interactions that would dramatically affect fauna. Effects of climate change on fauna are thus likely to be greater than indicated by these model outputs, especially where widespread tree loss eventuates. Indeed, given that changes in physical environments appear less significant for some groups, it may be changes in functional biological attributes that drive many faunal changes.

An overarching outcome of disruptions to a wide range of interactions in short timeframes is likely to be a simplification of ecosystems (including reduced diversity) at site and landscape scales. For example, dominance by species with traits typical of weedy species, for example, high dispersal capability and phenotypic plasticity, is not unlikely.

11 What conservation management options are available for TGE?

Ultimately, the outlook for TGE under predicted climate change scenarios is poor. Their existing environment is expected to be subject to moderate to high levels of stress under 2070 scenarios. In an undisturbed state, natural attributes of resilience might facilitate persistence of TGE species and structural characteristics, but forecasts become drastically more severe when past clearing and degradation is accounted for.

However, current land use trends and biodiversity conservation activity within the current extent of the TGE biome offer opportunities for improving the prospects of TGE and other native biodiversity. Accepting that the historical composition and structure of these ecosystems is unlikely to be fully maintained, we discuss five key conservation options in the context of the following overarching goals:

- Minimise extinctions of native species, at least at the Australian scale
- Where change is inevitable, promote reassembly by native rather than non-native species to maintain the evolutionary character of the Australian vegetation
- Optimise ecological functioning, including capture and use of available resources and maintenance of drivers of diversity (e.g. heterogeneity)
- Seek an appropriate balance between biodiversity conservation and human use.

We focus on lower risk approaches that are valuable independent of the extent of change that eventuates, that is, ‘no regrets’ options. These typically build on current conservation approaches that attempt to ameliorate non-climatic stressors and restore natural attributes of resilience in TGE, noting that these need to be applied with increased imperative and some adjustment of priorities.

11.1 Build on current approaches that offer a mix of on- and off-reserve protection, management and restoration

Consistent with past approaches, conserving suites of native species in fragmented TGE landscapes will continue to require appropriate management and protection of a diversity of sites that represent (1) natural environments and processes across the biome, and (2) different current and historical management regimes that result in differing biodiversity values. This is consistent with standard CAR conservation planning principles, but because of varied land tenures requires more complex planning and coordination (e.g. Lambert et al. 2000’ Prober et al. 2001).

Diverse remnants of TGE continue to offer the most cost-effective options towards this approach. Where these remnants do not persist in their current form under climate change, they are nevertheless likely to be the best local source of native species from which novel ecosystems may reassemble, and will provide the most appropriate sites and environments in which this can occur. Mechanisms for achieving these outcomes can build on considerable effort over the past 20 years to improve conservation of TGE, as discussed in section 5. Although similar mechanisms will be appropriate, new tools and approaches and some adjustment of management goals or ecological community circumscriptions may be necessary in due course to ensure they are sufficiently dynamic.

While this mix of approaches will remain relevant, we emphasise the following priorities that are increasingly relevant in a changing climate.

11.1.1 TARGET NATURAL SOIL ENVIRONMENTS

Native TGE soils are typically low in available macro-nutrients, contributing to the characteristic herbaceous flora (e.g. oligotrophic forbs) and associated fauna (e.g. grassland reptiles and birds), and conferring resistance to weed invasion. Sites retaining these soil qualities offer the greatest potential for conserving native species and promoting community reassembly by natives rather than exotics. Sites particularly relevant to this goal are those with a history of low grazing intensities and minimal fertiliser application. They include the extensive Travelling Stock Route network, unfertilised native pastures and woodland remnants on private land, and other rare sites that retain high native plant diversity and low weed cover (e.g. small, little-disturbed remnants in cemeteries).

11.1.2 IDENTIFY POTENTIAL LANDSCAPE AND REGIONAL REFUGIA

Some sites and regions are likely to promote the ongoing persistence of TGE species better than others owing to lower biotically scaled stress. Model outputs did not consistently predict key climate refugia, but levels of environmental stress were commonly lowest in Tasmania and central-western Victoria, and ANN suggested coastal woodland zones may be increasingly important for eucalypt woodlands. Regions such as the New England Tableland predicted to shift from forest and woodland to grassland could also act as refugia for fauna and flora dependent on the grassy ground layer.

At landscape scales, GDM outputs (Figure 7 and Figure 9) highlight the importance of areas with greater connectivity and topographic variation, which could facilitate local migrations to habitats retaining suitable environments. At the patch scale, this emphasises the importance of moisture-gaining areas low in the landscape. Because they are also the most productive parts of the landscape, most of these areas are already cleared or degraded, raising the significance of any that do retain significant diversity, woodland structure or unfertilised soils. Travelling Stock Routes (and their roadside equivalents in some states) traverse most parts of the landscape and are perhaps the single-most important source of such sites.

In addition to ‘places’, keystone habitat species can also define refugia. For example, summer flowering eucalypts are important seasonal refugia for birds, placing higher priority on managing for these species or resources over broad scales.

11.1.3 ALL ELSE BEING EQUAL, FAVOUR LARGE REMNANTS

Where management histories are similar, larger remnants are typically favoured because they support more species than smaller remnants, and have greater potential to withstand exotic invasion. This principle will be increasingly important under climate change. We emphasise, though, that native plant richness is more strongly related to management history than size, so small size alone should not preclude protection (e.g. Prober and Thiele 1995).

11.2 Tailor existing investments in restoration, revegetation and carbon sequestration towards biodiverse, climate-resilient outcomes

Efforts to restore site and landscape functioning in TGE already involve substantial revegetation and restoration initiatives. There is considerable potential to tailor these efforts towards climate resilient outcomes, at minimal additional cost. The same principle will apply to the forecast increases in woody plantings for carbon offsets associated with emerging emissions-trading policies. Notably, more so than in more intact landscapes, this offers significant opportunity to experiment with novel approaches to climate adaptation that elsewhere might be considered too great a risk to extant native diversity (Prober and Smith 2009). Techniques for achieving more climate-resilient outcomes are not well developed, but we suggest the following opportunities for further investigation.

11.2.1 PLANT DIVERSE MIXES OF LOCAL SPECIES

Inclusion of a broad mix of local species in plantings, especially those with distributions or traits that suggest tolerance of projected environments, is likely to increase the likelihood that a functioning native ecosystem will establish and persist. This is supported by GDM models that suggest about one in three local plant species could survive in 2070 environments. Diverse plantings might also target declining species with poor dispersal capacity, which are otherwise likely to survive into future environments. However, inclusion of these species should be informed by site conditions, as many declining species (e.g. geophytic forbs) are unlikely to perform well in weed-invaded or nutrient-enriched sites. Indeed, a greater proportion of remnants or native pastures should be targeted for this specific purpose.

11.2.2 EXPERIMENT WITH INTRODUCING GENES AND SPECIES APPROPRIATE TO THE DEGREE OF EXPECTED CHANGE

Historically, revegetation has focused on using local species and maintaining local genetic patterns of variation (Hufford and Mazer 2003). However, in a changing climate local genetic adaptations will by necessity be weakened (Pertoldi et al. 2007) and some species may not persist. A new framework is thus needed to guide use of genetic material towards climate-resilient restoration and revegetation. We propose that from an ecosystem- or site-based perspective, an appropriate approach is to increase the degree of risk taken regarding introduction of novel genetic material with increasing degrees of threat or change. This could include:

1. Under low expected change, use local species and local genes
2. Where persistence of a local species is questionable, test whether its capacity to persist can be enhanced through inclusion of more resilient genotypes in plantings; that is, use local species but intra-specific variation from a wider variety of sources or a source reflecting likely future environments
3. Where few local native species are expected to persist, use novel species, first from close by (that may have the capacity to migrate to the site naturally), then species from increasingly distant locations with increasing degrees of threat.

The latter is a high risk option due to uncertainties in predictions and potential impacts on local species, although we note that non-local species are already commonly used in wheatbelt plantings (e.g. favouring species more resistant to salinisation).

From a species-based perspective, these recommendations would translate to:

1. Re-introduce to sites where they are likely to once have occurred (and conditions are suitable), using the most local healthy genetic sources
2. Attempt to enhance capacity of re-introductions to adapt in ‘native’ sites by augmenting or selecting for appropriate natural genetic diversity within species
3. For highly threatened species, consider areas of likely suitable future environments, beginning within the local landscape and moving outwards.

11.2.3 VARY LEVELS OF CONNECTIVITY

A common goal of revegetation is to restore landscape connectivity to promote persistence of woodland fauna (Doerr et al. 2010). This would similarly restore resilience to climate change by facilitating species migrations and promoting adaptive ecosystem change to maintain optimal functioning. On the other hand, it could expedite replacement of local native species by exotic or native species from elsewhere (Dunlop and Brown 2008). At the broad scale, planting designs might thus aim to provide areas of low and high landscape permeability. The latter could focus on corridors aligned with climatic gradients, such as already underway in the ‘Kosciuszko to Coast’ landscape connectivity project.

11.2.4 REDUCE NON-CLIMATIC STRESSORS

Soil compaction and carbon depletion in many degraded TGE have potentially compromised soil–water relations, and nutrient enrichment has increased competition with invasives (Prober et al. 2002b; McIntyre and Lavorel 2007). Restoration of these processes to maximise capture of declining rainfall or increase resistance to nutrient-loving exotics could thus improve resilience of plantings and degraded remnants under climate change. Similar principles apply to other compromised processes such as those leading to tree decline, and associated loss of resources for fauna.

11.3 Favour land use changes with positive rather than negative biodiversity outcomes

As discussed above, revegetation and carbon sequestration initiatives offer potential advantages for TGE biota under climate change. Other land use trends likely under climate change, including extensification (e.g. reversion to low-input native pastures) and abandonment, offer similar potential advantages. Indeed, increasing the area of native pastures could be more important than plantings for improving the prospects of native herbaceous species tolerant of livestock grazing, and is already supported under schemes such as the box-gum grassy woodlands stewardship program.

By contrast, some land use changes will exacerbate the negative outcomes of climate change for biodiversity. These include replacement of low intensity land uses with more intensive agriculture, such as irrigated vineyards or biofuel crops. As well, plantings for carbon sequestration could favour woody rather than grassy communities. Policies attempting to moderate land use change will thus need to balance the social and economic benefits with biodiversity impacts.

11.4 Manage exotic species

Novel communities dominated by exotic species are a likely outcome of climate change in many woodland remnants in the absence of exotic species management. Strengthening of current approaches to exotic management is needed to minimise these impacts. These include quarantine, identification and control of sleeper (incipient) exotics, containment of incipient transformer exotics (particularly buffel grass), as well as more intensive weed and feral control in selected conservation assets.

Notably, some of the most pervasive existing weeds could be disadvantaged by declining cool-season precipitation. This could provide an opportunity to manipulate recolonisation of selected, currently invaded sites by native rather than exotic species (e.g. by augmenting native propagules).

11.5 Manage disturbance to maintain open grassy ecosystems where appropriate

Given the general human desirability of open woodland landscapes (Balling and Falk 1982) and the poor outlook for TGE elsewhere in south-eastern Australia, it may be desirable to manage some areas to promote persistence of the open, grassy structure of TGE where there is increasing pressure of woody thickening. Shrub-grass dynamics in semi-arid ecosystems are relatively well-understood, and it is likely that they can be manipulated in some areas to favour open grassy ecosystems (Lunt et al. 2012). In particular, fire may be an effective management tool in areas predicted to become more suitable for *Callitris* forest.

12 Conclusions

Although uncertainties are considerable, climate change in the order of 2070 medium and high scenarios is likely to dramatically impact on TGE, exacerbated by interactions with current degradation. While it is not likely to be possible or desirable to entirely prevent ecological change, it is possible to ameliorate non-climatic limits to adaptation and promote reassembly by native rather than exotic species. Current conservation and restoration efforts already target similar goals, and reinforcing and adjusting these approaches offer the highest priority, lowest risk climate adaptation options in the degraded TGE biome. Notably, because natural integrity is already highly compromised, this may facilitate experimentation with higher risk adaptation options, such as genetic enhancement of populations, fire management, facilitation of local landscape shifts, and species translocations.

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Appendix A Participants in the temperate grasslands and grassy ecosystems climate change workshop, April 2009

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Justin Billings	Department of Environment, Water, Heritage and Arts
Ross Bradstock	University of Wollongong
Kerry Bridle	CSIRO Sustainable Ecosystems/University of Tasmania
Sue Briggs	Department of Environment, Climate Change and Water, NSW
Don Butler	Macquarie University, Environment Protection Authority, Qld
Oberon Carter	Department of Primary Industries and Water, Tasmania
Saul Cunningham	CSIRO Entomology
Liz Dovey	Department of Climate Change
Angela Duffy	Department of Environment and Heritage, SA
Michael Dunlop	CSIRO Sustainable Ecosystems
Simon Ferrier	CSIRO Entomology
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David Keith	Department of Environment, Climate Change and Water, NSW
Ian Lunt	Charles Sturt University
Sue McIntyre	CSIRO Sustainable Ecosystems
Tim Milne	Nature Conservation Society, SA
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