Scenarios for future biodiversity loss due to multiple drivers reveal conflict between mitigating climate change and preserving biodiversity

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Abstract
We assess the potential for future biodiversity loss due to three interacting factors: energy withdrawal from ecosystems due to biomass harvest, habitat loss due to land-use change, and climate change. We develop four scenarios to 2050 with different combinations of high or low agricultural efficiency and high or low meat diets, and use species–energy and species–area relationships to estimate their effects on biodiversity. In our scenarios, natural ecosystems are protected except when additional land is necessary to fulfil the increasing dietary demands of the global population. Biomass energy with carbon capture and storage (BECCS) is used as a means of carbon dioxide removal (CDR) from the atmosphere (and offsetting fossil fuel emissions). BECCS is based on waste biomass, with the addition of bio-energy crops only when already managed land is no longer needed for food production.

Forecast biodiversity loss from natural biomes increases by more than a factor of five in going from high to low agricultural efficiency scenarios, due to destruction of productive habitats by the expansion of pasture. Biodiversity loss from energy withdrawal on managed land varies by a factor of two across the scenarios. Biodiversity loss due to climate change varies only modestly across the scenarios. Climate change is lowest in the ‘low meat high efficiency’ scenario, in which by 2050 around 660 million hectares of pasture are converted to biomass plantation that is used for BECCS. However, the resulting withdrawal of energy from managed ecosystems has a large negative impact on biodiversity. Although the effects of energy withdrawal and climate change on biodiversity cannot be directly compared, this suggests that using bio-energy to tackle climate change in order to limit biodiversity loss could instead have the opposite effect.

Keywords: biodiversity loss, climate change, biomass harvest, land use, biomass energy, carbon dioxide removal

1. Introduction
Biodiversity is inextricably linked with human wellbeing; through its contribution to the functioning and resilience of ecosystems; as a resource with the potential for discovery of new compounds or processes; and a source of emotional wellbeing (Díaz et al 2006). Natural ecosystems provide resources including food, fresh water, fibre etc, and biodiversity loss affects the ability of ecosystems to fulfil these roles. This in turn affects poorest communities the most, since they are least able to afford to substitute the lost ecosystem services.
At the same time, access to proper nutrition is fundamental to wellbeing. Around 1 billion people are currently in food poverty, and the global population is forecast to grow from the current ∼7 billion to ∼9.3 billion people in 2050. Providing everyone with an adequate diet is seen as one of the greatest challenges for human wellbeing in the coming decades. Yet the loss and fragmentation of natural habitats due to agricultural expansion has been a major cause of biodiversity loss to date.

Climate change also affects human wellbeing, for example through more extreme weather events, changing distributions of disease vectors, and forced migration, as well as being a key driver of biodiversity loss. Climate change is already removing unique habitats and contracting others faster than some species can disperse, and it will become an increasingly important driver of biodiversity loss in the future (Thomas et al 2004a); indeed one of the key motivations for tackling climate change is to protect natural ecosystems and the goods and services they provide.

The joint pressures of climate change and expansion of agriculture set up a potential conflict between elements of human wellbeing; meeting food demand and improving diets requires the growth of agriculture, which in turn causes biodiversity loss and contributes to climate change. The use of bio-energy crops, in particular, is put in a highly ambiguous position among these interactions. They have the potential to mitigate climate change, but may also compete with food production, cause the destruction of natural habitats, and even cause CO₂ emissions as a result of land-use change.

Here we take an integrated view of global terrestrial biodiversity loss, seeking to quantify the multiple effects of different future land-use scenarios, and examining the interactions between different drivers. In particular, agriculture withdraws energy from ecosystems to feed us, is by far the largest driver of land-use change, and contributes around 30% of greenhouse gas emissions at present. Our scenarios (Powell and Lenton 2012) are driven by increasing population and calorific intake, changing dietary demand for animal products, changes in the efficiency of food production, and a drive to use biomass to mitigate climate change, where it does not conflict with food production or the preservation of remaining natural ecosystems. In particular, we consider the potential for biomass energy with carbon capture and storage (BECCS) to remove carbon dioxide from the atmosphere and offset fossil fuel emissions. We focus on the use of waste biomass for BECCS, only allowing dedicated bio-energy crops where land becomes abandoned from agriculture.

Our approach to estimating biodiversity loss follows previous work in using concepts from ecological theory, namely the species–area and species–energy relationships. The species–area relationship is a classic tenet of ecology describing a (non-linear) correlation between increasing area and the number of species to be found. The converse formula has been widely used to predict extinction rates from habitat loss (Pimm and Raven 2000). Estimates of biodiversity loss from range shifts due to climate change also make use of the species–area relationship (Bakkenes et al 2002, Thuiller 2003), although only one study has attempted a quantitative global assessment (Thomas et al 2004a).

Less widely recognized is that the withdrawal of energy from ecosystems, through the harvesting of biomass—for food, fibre, wood products or energy—also causes biodiversity loss. The corresponding species–energy relationship describes the correlation between species diversity and the energy available to organisms at a given spatial resolution (Wright 1983, Gaston 2000). While at a local scale the very highest levels of energy availability may lead to competitive dominance of a relatively low number of species causing a peaked distribution, at larger spatial scales there is generally a monotonic positive correlation between energy availability and diversity (Chase and Leibold 2002, Evans et al 2005, Mittelbach et al 2001).

The species–energy relationship appears to be driven by high species occupancy at higher energy levels, with greater availability of energy allowing greater coexistence of species by providing a wider range and complexity of niches in space, time and in the number of possible community compositions (Chase and Leibold 2002, Bonn et al 2004, Jetz and Fine 2012). Higher productivity may also lead to more species by increasing the probability of occurrence of resources that enable the persistence of viable populations (Storch et al 2005). Species that occupy lower energy levels tend to be generalists with large ranges that are also present when more energy is available (except, of course, for those that specialize in extremely low energy environments), while higher energy levels support a greater range of specialists with small ranges (Bonn et al 2004). Higher energy biomes, e.g. in the tropics, are therefore more sensitive to reductions in energy availability.

The withdrawal of energy from ecosystems can be quantified in terms of the human appropriation of net primary production (HANPP). This is defined as the combined effects of anthropogenic changes in productivity, and harvest of biomass, on the availability of NPP in ecosystems (Haberl et al 2007). HANPP affects biodiversity via the species–energy relationship, since the anthropogenic removal of NPP constitutes a reduction in the energy available to other organisms.

Positive correlations have frequently been observed between intensity of agriculture and biodiversity loss, with the drivers being a combination of landscape effects, intensity of inputs and intensity of biomass extraction (Eaglington and Pearce-Higgins 2012, Kleijn et al 2009, Haberl et al 2005). These drivers are not independent of one another, and as such HANPP as well as being a cause of biodiversity loss in itself, may be considered a proxy for the other elements of agriculture which have a negative impact on biodiversity, including excessive nutrient application, chemical pollution and the creation of relatively homogeneous agro-ecosystems.

Very few studies have however characterized the impact of agriculture on biodiversity in terms of HANPP and the species–energy relationship (Haberl et al 2004, 2005, Wright 1990). Among these are observational studies of the relationship in several phylogenetic groups on Austrian farmland (Haberl et al 2004, 2005). One pioneering study used a generalized species–energy relationship to predict global species endangerment from 1990 to 2000 based on
expected increase in food demand due to population growth (Wright 1990), but did not include dietary trends.

Extant flows of biomass energy can be interpreted in terms of demand driven by population and demographics (Krausmann et al 2008), and future demand can be forecast (Powell and Lenton 2012). Such material- and energy-flows analyses are powerful tools for the description of human impact on ecological energy and carbon fluxes, linked to the concept of a socio-ecological metabolism (Fischer-Kowalski and Haberl 2007, Haberl 2006). This framework describes the physical relationship between human society and our environment in terms of inputs and outputs of energy and resources, and provides a common accounting method for the intimately linked carbon and energy cycles of humanity and the Earth system as a whole.

2. Methods

2.1. Biomass flows model

We use a simple model of biomass flows and land use in the global agricultural system, developed from a previous study of the carbon dioxide removal (CDR) potential from biomass energy generation with carbon capture and storage (BECCS) (Powell and Lenton 2012). The model uses the predicted growth in human population to 9.3 bn in 2050 combined with an expected rise in average daily calorific consumption of ~20% (Smeets et al 2007) to forecast the increase in global food demand to 2050. Conversion factors derived from literature sources (Krausmann et al 2008, Wirsenius 2003) and the FAOSTAT database are used to calculate the biomass harvest required to meet food demand each year, from a combination of primary crops and livestock fed by grazing or from market feed. Projected biomass harvest for fibre and forestry products are also included, the former driven by population growth and the latter based on FAO projections (FAO 2010).

The demand for biomass harvest is combined with average yield data for primary crops, fodder crops and pasture, as well as non-food products, to calculate the land area required for each. Average crop yields are assumed to increase 1% annually, which in this biomass flows approach could be met either by increasing actual yields, or by reduction of the ‘yield gap’ that exists due to environmental and management factors. A 10% increase in stocking intensity on pasture is also assumed between 2000 and 2050. We assume no increase in yields from forestry.

Where possible, expansion of increasing land-use types is met by decreases in land use of other sectors, with a hierarchy of allocation from food crops down to bio-energy crops. If, in a given time-step, the overall area under management is required to expand (figure 1), land is appropriated from natural ecosystems. These are divided into five classes varying in carbon stocks based on IPCC guidelines for greenhouse gas inventories, and on data for above-ground net primary productivity (NPP) (Haberl et al 2007). These five biome groups approximately equate to; tropical rainforest; deciduous forest; grassland and savanna; boreal forest; and desert and tundra (table 1). Expansion of croplands displaces pasture, which in turn appropriates land from the three most productive categories (tropical rainforest; temperate deciduous forest; grassland and savanna) in a ratio of 1:2:2, as long as 15% of each class is preserved. Managed forests also expand into natural biomes unless accommodated by shrinkage of other land uses. Excess land requirement is taken up by the other classes or if necessary the next most productive unused class. This reflects the historical tendency for agricultural land to occupy the most productive land types (Ramankutty and Foley 1999). Indeed 80% of the expansion of cropland since the 1980s has been in the tropics (Gibbs et al 2010).

Of the four scenarios (table 2), two meet the expected rise in the per-capita consumption of animal products from 16.06% of daily energetic intake in 2000 to 18.8% in 2050 (Smeets et al 2007). Two alternative ‘low meat’ scenarios imagine a deliberate, or enforced, reduction in consumption of animal products to 15% of daily calorific intake by 2050. The
scenarios are further separated into high and low agricultural efficiency variants. The high efficiency scenarios follow trends in the livestock system towards a greater contribution from pig meat, poultry and eggs, which are up to a factor of ten more efficient at converting primary biomass to food than are ruminants (Wirsenius 2003). Indeed they are even more efficient in terms of land area since they can be fed on high quality feed crops rather than from extensive pasture (although this means that pigs and poultry compete directly with humans for primary crops). These scenarios are further focused on efficiency of land use by an increase in the use of fodder crops rather than pasture as feed for ruminants, representing intensification of ruminant farming systems. This in turn allows a higher proportion of collection and recycling of manure. High efficiency scenarios also see increasing recycling of residues and reduction of food waste. In the low agricultural efficiency scenarios, efficiency gains slow after 2010 and cease altogether after 2015.

In each of the scenarios, waste biomass streams are made available as feedstocks for BECCS processes, which offset fossil fuel CO₂ emissions as well as actively generating net negative carbon fluxes. All scenarios see an initial increase in land needed to meet food demand, but in those that see a subsequent decline in land-use area any land abandoned by food production is considered suitable for growing lignocellulosic biomass crops, which are also fed into BECCS schemes. Perennial lignocellulosic crops such as Miscanthus sp. have been shown to produce viable yields even on low quality, degraded land (Liu et al 2012). In all of the scenarios here, BECCS activity begins in 2010 and increases to full capacity in 2030.

The four scenarios thus include three drivers of terrestrial biodiversity loss; biomass harvest; conversion of natural habitats; and climate change, which we now detail.

### 2.2. Biomass harvest on managed land

Here we use %HANPP, defined as the harvested biomass as a percentage of the above-ground NPP of the potential vegetation before human influence (NPP₀), to predict the effect of intensifying and expanding agriculture on species richness. At each yearly time-step, %HANPP is calculated for each of seven main land-use types on actively managed land, and their relative areas are used to calculate a weighted average for %HANPP on managed land (figure 2). This is thus affected by the intensity of agriculture (which tends to increase as yields rise and stocking densities increase), the relative demand for the different land uses, and the NPP₀ of the land under use.

As land-use demand changes within the categories of land already under management, managed land is redistributed according to an order of priority with food crops the highest and bio-energy crops the lowest. Bio-energy crops are only allowed when overall human land use would otherwise decrease, i.e. land is being abandoned from food production. In some cases, the local intensity of biomass harvest decreases, for example if land used for growing crops is converted to pasture. Such decreases in %HANPP cannot lead to net increases in biodiversity, since an increase in global biodiversity can only be caused by speciation events. However, local easing of biomass harvest is considered to provide a buffer for loss of diversity elsewhere, by providing potential habitat for displaced species, and as such is included in the weighted average.

A species–energy relationship is used to estimate the effect of %HANPP on biodiversity. This is derived from an empirical study of the relationship between HANPP and biodiversity on managed lands in Austria (Haberl et al 2004). This study is to our knowledge the only attempt to quantify the relationship in terms of HANPP, and does so across croplands,
pasture, managed forests and urban environments, and across multiple taxonomic groups. Species richness (\(S\)) declines with \%HANPP according to:

\[
\log(S) = a + b \cdot \log(\%HANPP)
\]  

(1)

where \(a\) is a constant, \(b = -1.6\) for autotroph diversity and \(b = -1.1\) for heterotroph diversity. We assume equal weighting of autotrophs and heterotrophs, combining them to give a total average effect on biodiversity. We note that species–energy curves can vary significantly among taxonomic groups, biota and spatial scales, and as such these curves derived from typical Austrian flora and fauna may not be representative of those across the world’s agricultural systems.

2.3. Habitat loss via land-use change

The species–area relationship is used to determine the proportion of affected species on each natural land class as agriculture expands:

\[
S = cA^z
\]  

(2)

where \(S\) is species richness, \(c\) is a constant and \(z = 0.25\) (Rosenzweig 1995, Thomas et al 2004a, Pimm and Raven 2000). A weighted average for species loss from natural ecosystems is produced according to their relative areas (\(A\)), and their differing species richness. Here we use a typical species–energy curve (Rosenzweig 1995, Wright 1990) to estimate the natural gradient in species diversity from less productive to more productive biomes:

\[
S = dE^z
\]  

(3)

where \(S\) is species richness, \(d\) is a constant, \(E\) is energy taken to be NPP, and \(z = 0.5\), reflecting the steeper curves usually associated with continental to global-scale trends (Wright 1983, Mittelbach et al 2001). This gives around an eight-fold variation in diversity between the most and least productive biome types.

Since this area is not destroyed as agriculture expands, merely transferred to a managed land-use type, expansion could be assumed to cause a corresponding increase in the species represented on managed land via the species–area relationship. We assume, however, that since the species present in low energy habitats tend to be generalists also found in high energy habitats (Bonn et al 2004), any species surviving the transition are likely to be generalist species already present in managed environments. Our method for estimating global species loss shows low sensitivity to this assumption.

In addition to the area of habitat lost, we also account for the effect of differences in the productivity of appropriated land. At each time-step the remaining areas of each natural land class are used to calculate a weighted average NPP for unmanaged land. The resulting negative trend in remaining natural NPP, as high productivity land is favoured for agriculture, is then treated as further HANPP, and a species–energy curve applied to determine the associated species loss.

Habitat loss is thus defined by changes in area of habitat types, and the relative diversity of biomes in which habitat loss occurs.

2.4. Climate change

To calculate the consequences of our four scenarios on atmospheric CO\(_2\) and climate change, we use a simple Earth system model (Lenton 2000, Vaughan and Lenton 2012). The model is forced from 1800 to year 2000 with historical estimates of fossil fuel emissions (Marland et al 2008) and land-use change emissions (Houghton 2008), predicting 369.4 ppm CO\(_2\) in year 2000 and global warming of 0.89°C (from 1800) in good agreement with observations (Vaughan and Lenton 2012).

From 2000 onwards, atmospheric CO\(_2\) and global temperature change are determined by the combination of a common ‘baseline’ fossil fuel emissions scenario to which each of our four scenarios are added. Although absolute CO\(_2\) and temperature change depends on the choice of baseline future fossil fuel emissions scenario, deviations from that baseline due to the scenarios are insensitive to the choice of baseline—as expected from theory (Lenton and Vaughan 2009).

For our ‘baseline’ fossil fuel emissions after 2000 we follow an existing mitigation scenario ( Vaughan and Lenton 2012), using estimated fossil fuel (plus cement production) emissions for 2000–2005 (Marland et al 2008), followed by a 1.7% yr\(^{-1}\) increase from 2005 to 2015 (the long term mean growth rate over the last 25 years), after which mitigation activity begins in earnest and it takes 40 years to transition to a 1.7% yr\(^{-1}\) decrease in emissions. This scenario gives peak fossil fuel emissions of 11.35 PgC yr\(^{-1}\) in 2035, declining to 10.3 PgC yr\(^{-1}\) in 2050.

Each scenario has contributions to the global CO\(_2\) balance from land-use change emissions, carbon dioxide removal (CDR), and offsets of fossil fuel emissions by bio-energy. These three components are added together to get an overall CO\(_2\) flux, either to or from the atmosphere, at each
Figure 3. Calculated changes in (a) atmospheric CO$_2$ concentration, and (b) global temperature under our four scenarios, together with the baseline changes due to fossil fuel emissions only, which are the same in all four scenarios.

time-step. CO$_2$ emissions from land-use change are calculated using the IPCC tier 1 methodology, according to the carbon stocks of the vegetation on each land class and the land use replacing it (IPCC 2006). CDR and offsets are calculated as described elsewhere (Powell and Lenton 2012).

The combined flux is initially dominated by land-use change and is therefore a net CO$_2$ source to the atmosphere. It is identical in all four scenarios up to 2010, declining from 1.77 PgC yr$^{-1}$ in 2000 to 1.27 PgC yr$^{-1}$ in 2010, which is consistent with estimates that land-use change emissions declined markedly over that decade (Friedlingstein et al 2010). Our land-use change emissions are above the estimated mean but well within the error range (Friedlingstein et al 2010). In previous work (Vaughan and Lenton 2012), a lower estimate of land-use change emissions was used for 2000–2005 (Houghton 2008), but the absolute values agree well in 2005.

Atmospheric CO$_2$ varies by $\sim$50 ppm in 2050 from 452 ppm in the low meat high efficiency scenario to 498 ppm in the high meat low efficiency scenario (figure 3(a)). The corresponding global temperature range in 2050 is 0.29 °C, from 0.89 °C to 1.18 °C warming above 2000 (figure 3(b)).

We base the sensitivity of terrestrial biodiversity to climate change on previous results (Thomas et al 2004a), which use bioclimatic modelling of range shifts coupled with the species–area relationship to estimate overall species committed to extinction under a number of climate model projections. We fit a linear regression through their results, having found no significant difference between the fit of linear and polynomial models, producing a sensitivity of around 12.6% species loss per degree of climate change. This is between their two scenarios of ‘no dispersal’ and ‘full dispersal’, and we note that in reality dispersal is likely to be limited by anthropogenic land use and habitat fragmentation, but is unlikely to be prevented entirely. We then use this value to forecast loss of terrestrial biodiversity due to climate change relative to the year 2000.

2.5. Sensitivity analysis

To analyse the sensitivity of our results to variation in the strength of the relationships driving biodiversity loss, we tested the effect of varying the key coefficients in each relationship on the final biodiversity loss caused by the relevant driver. In the case of the species–energy relationship on managed land (equation (1)) and the species–area relationship (equation (2)) this meant varying the pertinent coefficients ($b$ or $z$) up to $\pm$25% from the value used in our study, approximately spanning the range commonly found in the literature (Rosenzweig 1995, Wright 1990, Pimm and Raven 2000, Thomas et al 2004a). Since the gradient of the species–energy relationship appears to increase over larger spatial scales, and we were concerned about under-estimating the range in diversity across biomes, we varied $z$ in equation (3) between 0.25 and 1.5, producing up to 100 fold differences in diversity between the most and least productive natural biomes. For the effect of climate change, sensitivity of the results was measured for a variation of $\pm$50% of the climate change sensitivity coefficient, spanning the range of outcomes given by Thomas et al (2004a). We thus obtained maximum and minimum variants for our forecasts of biodiversity loss from each scenario.

3. Results

We divide our forecasts of committed terrestrial biodiversity loss in 2050 from 2000 levels into three components (figure 4): the effects of energy withdrawal due to biomass harvest on managed land, the effects of habitat loss due to land-use change from natural biomes, and the effects of climate change on all land types. Whilst the first two contributors to biodiversity loss could in theory be added up (because they refer to mutually exclusive fractions of the land surface) we have not done so as we are not able to appropriately weight the differences in diversity between managed land and natural biomes. Climate change as a cause of biodiversity loss cannot be considered additive to the other two contributors, because the same species may be vulnerable to both climate change and either energy withdrawal from managed lands or habitat loss from natural biomes. Conversely there are potential synergies between the effects of biomass harvest, habitat loss and climate change,
which could make their effects on biodiversity greater than additive (Sala et al. 2000, Brook et al. 2008, Eglington and Pearce-Higgins 2012).

Looking across our four scenarios, using our default parameter settings, forecast committed biodiversity loss in 2050 due to habitat loss from natural biomes ranges over 1–20% (figure 4(a)), that due to biomass harvest on managed land ranges over 19–38% (figure 4(b)), and that due to climate change on all land types ranges over 11–15% (figure 4(c)). Thus, the effect of variation across the four scenarios on biodiversity loss is largest for habitat loss due to land-use change and smallest for climate change. The effect of habitat loss on biodiversity (figure 4(a)) is largest in the high meat, low efficiency scenario and smallest in the low meat, high efficiency scenario. However, the effect of energy withdrawal on biodiversity (figure 4(b)) is greatest in the low meat, high efficiency scenario and comparable in the other three scenarios. The effect of climate change on biodiversity (figure 4(c)) is greatest in the high meat, low efficiency scenario and smallest in the low meat, high efficiency scenario, but variation between the scenarios is low.

Since the results for managed land and natural biomes refer to mutually exclusive sets of species, with potentially very different levels of diversity, and results for climate change driven biodiversity loss refer to all terrestrial species, no quantitative comparison can be made between the three sets of results. It is clear however, that 1% loss of diversity from productive natural biomes represents a higher proportion of global biodiversity than the equivalent loss on human-dominated land. Furthermore, species at risk from climate change are likely to be specialized organisms with narrow niche-spaces, and therefore likely live in higher energy natural biomes (Storch et al. 2005, Thomas et al. 2004a), making biodiversity loss from natural ecosystems of particular significance.

Despite being unable to directly compare the consequences of the three drivers of biodiversity loss, the capacity of the four different scenarios to affect biodiversity in different ways is clear, and the differing ranges and patterns of each set of results allows us to draw some conclusions.

There is a clear difference in effect on habitat loss between our high and low efficiency scenarios, with biodiversity loss from natural biomes of 12.0–20.0% in low efficiency scenarios and only 1.3–2.8% in high efficiency scenarios (figure 4(a)). This is driven by the huge requirement for land in the low intensity, low efficiency livestock systems, which appropriate vast areas of productive natural biomes, leading to an increase in total human land use from 5.17 Gha in 2000 to 8.45 Gha in the high meat variant and 7.08 Gha with a ‘low meat’ diet by 2050 (figure 1). The preferential use of more productive ecosystem types for the expansion of agriculture increases this pressure, indeed in the high meat, low efficiency scenario the two most productive land classes are reduced as far as allowed in the model, leaving only 15% of their year 2000 area standing. In the high efficiency scenarios, very little growth in agricultural area is required, human land use reaching its maximum in 2034 at 5.70 Gha under the forecast ‘high meat’ diet and 5.46 Gha in 2014 under ‘low meat’ (figure 1). As a consequence, biodiversity loss from natural biomes is low. These results are clearly sensitive to the parameters assumed for the relationships used to produce them, the largest variations in the sensitivity analysis producing 2–3.5 fold variation in estimates of biodiversity loss—however, the effect of habitat loss on biodiversity if agricultural efficiency cannot be increased in future still stands out.
The intensification of biomass harvest required by high efficiency agriculture may carry some biodiversity cost (figure 4(b)), but this is relatively small. All scenarios see a significant loss of diversity from agricultural land due to increasing intensity of harvests from 2000 to 2050. The ‘high meat, low efficiency’ sees a biodiversity loss of 21.5%; ‘low meat, low efficiency’ a loss of 19.4%; and ‘high meat, high efficiency’ a loss of 23.1%, due to withdrawal of energy on managed land. Biodiversity losses are somewhat lower in the low efficiency scenarios, because although they require a huge land area the bulk of this is made up of pasture with relatively low biomass removal and relatively high diversity, leading to %HANPP of around 25–30%, as opposed to 65–85% for the fodder crops used more extensively in high efficiency scenarios. Increased use of fodder crops and higher stocking intensities in the high efficiency scenarios mean that, although biomass harvest is lower overall—as a result of reduced contribution of inefficient ruminants to livestock products, greater recycling and reduced food waste—it is more concentrated, with average %HANPP on managed land of 43.2% by 2050 in the ‘high meat, high efficiency’ scenario. However, the resulting increase in biodiversity loss on managed land is modest when compared to the huge reduction in species loss from natural biomes caused by this concentration of farming. By far the largest biodiversity loss due to biomass harvest occurs in the ‘low meat, high efficiency’ scenario because the roughly 660 Mha reduction in land required for food production allows for the conversion of pasture to a correspondingly large swathe of bio-energy plantation. The increase in %HANPP on this area from around 25% for pasture to almost 90% for bio-energy plantation contributes to a 38.3% species loss from managed land in this scenario.

The effect of climate change on biodiversity loss varies least across the four scenarios (figure 4(c)) primarily because the same baseline climate change of $\sim-1^\circ$C from 2000 to 2050 due to fossil fuel emissions occurs in all four scenarios. There is only a $\sim0.3^\circ$C range in 2050 global warming across the scenarios. Climate change is least in the ‘low meat, high efficiency’ scenario thanks largely to the aforementioned bio-energy plantations, supporting a net carbon dioxide removal flux of $\sim5$ PgC yr$^{-1}$ in 2050. However, inertia in the carbon cycle and the climate system means that the full climatic effect of this activity would not be felt until later. Climate change is greatest in the ‘high meat, low efficiency’ scenario, due primarily to CO$_2$ emissions from land-use change, however even in this scenario the conversion of biomass wastes to stored carbon is able to reduce the net flux of CO$_2$ from the land surface to close to zero in 2050.

The absolute percentage values of biodiversity loss forecast here should be viewed as highly uncertain, with many caveats accompanying them. The globalization of species–area or species–energy relationships has many potential flaws, several of which are highlighted by previous debate surrounding the estimated effect of climate change on biodiversity loss (Thuiller et al 2004, Buckley and Roughgarden 2004, Harte et al 2004, Thomas et al 2004b). Possible interactions between the species–area and species–energy relationships are not considered (Storch et al 2005). Potential synergies between the effects of habitat loss and fragmentation, biomass harvest and climate change on biodiversity are ignored (Sala et al 2000, Brook et al 2008, Eglington and Pearce-Higgins 2012). Our approach also fails to account for diverse management strategies according to regions, cultures, socio-economic conditions and history (Erb et al 2012).

Despite these caveats, the relative effects of different drivers on biodiversity loss include some striking results that are robust to our sensitivity analysis. In particular, using bio-energy with carbon capture and storage to tackle climate change is likely to have a greater negative effect on biodiversity due to energy withdrawal from ecosystems, than the avoided biodiversity loss due to less climate change.

4. Conclusion

Although the absolute figures are highly uncertain, we are able to draw some tentative conclusions about the relative importance of different drivers of biodiversity loss. First, if current trends of increasing agricultural efficiency and intensification are not maintained, meeting food demand by expanding agricultural land area could become the dominant cause of future biodiversity loss, via the destruction of productive, and therefore species rich, natural habitat, with additional losses from the harvesting of biomass energy and the climatic consequences of land-use change CO$_2$ emissions. Second, although continued increases in agricultural efficiency could liberate land for dedicated bio-energy crops, the resulting withdrawal of energy from managed land would likely have a much greater negative impact on biodiversity than the positive effect of reducing climate change. Clearly biodiversity loss is not the sole reason for mitigating climate change, but it is a significant one, so this result highlights a potentially important contradiction implicit in many climate change mitigation strategies.

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