**Working title: Positive feedbacks cause collapse of tree cover in a temperate forest**

Main message of paper: Transition to non-forest state depends on positive feedbacks: loss of mature trees causes lower recruitment and tree death increases likelihood of death of nearby trees

Authors: Philip Martin1, Adrian C. Newton1, Elena Cantarello1, Paul Evans1, Edward Mountford2.

Journal: Journal of Applied Ecology

1Centre for Conservation Ecology and Environmental Sciences, Faculty of Science and Technology, Bournemouth University, Poole, BH12 5BB, UK.

2Joint Nature Conservation Committee, Monkstone House, City Road, Peterborough, PE1 1JY, UK.

Keywords: forest dieback; regime shift; forest collapse; ecosystem resilience;

Corresponding author email: phil.martin.research@gmail.com

**Summary**

1. There is concern that forest dieback may lead to transitions to non-forest states. Positive feedbacks are thought to play an important role in such transitions by creating self-perpetuating shifts in system states. Despite this concern, there is has been relatively little work on the topic.
2. We used statistical models to identify correlates of death and recruitment of the canopy dominant (*Fagus sylvatica* - beech) in a temperate forest site that has been sampled over 50 years and appears to be undergoing transition to grassland. We use these results along with information from the literature to build an individual based model to investigate the impact of positive feedbacks on forest persistence.
3. We found that the probability of tree death declined with increasing tree growth rate, distance to nearest dead tree and sand content of soils, but positively correlated with tree size. Seedling density was negatively related to canopy cover, but sapling density was higher in areas of high canopy cover.
4. Our individual based model showed that when juvenile survival in treeless areas was reduced and mature trees were more likely to die when close to other dead trees forest structure collapsed. Otherwise forest structure remained stable.
5. Our results suggest that positive feedbacks that influence both recruitment of juveniles and death of mature trees are likely to be partly responsible for a transition to non-forest state in our study site.This result was contingent on the base mortality rate of juvenile trees being relatively high, which is likely to be the case in our study area which has a high density of deer and ponies.
6. **Synthesis and applications:** To enhance forest resilience management should attempt to stop the development of such positive feedback loops. In our study area fencing off forest areas to reduce seedling mortality caused by browsing of ponies and deer may reduce feedbacks related to juvenile death. However, reducing the feedbacks related to mortality of larger trees will be more challenging as these may be related to larger scale drivers relating to climate change.

**Introduction (aim for ~600-800 words)**

There is widespread concern that forest dieback may result in shifts to non-forest states (Reyer et al., 2015). Over the past decade research has suggested such shifts may occur in both tropical (Barlow and Peres, 2008; Hirota et al., 2011) and boreal regions (Scheffer et al., 2012) as a result of a both changes in climate and disturbance regimes. Forests may be particularly vulnerable to rapid changes because they trees are long lived, immobile organism that consequently find it difficult to adapt to new environmental conditions (Burrows et al., 2011; Seidl et al., 2015). Any shift to relatively treeless, non-forest states would cause in loss of forest biodiversity as well dramatic changes in the ecosystem services provision (Scholes et al., 2014). Due to these risks a recent IPCC assessment concluded that forest dieback has the potential to cause major economic impacts (Scholes et al., 2014). However, despite these concerns there is relatively little known about the mechanisms that may forests to transition to non-forest states (Reyer et al., 2015).

Transitions from one ecosystem state to another occur when perturbations result in changes to a system from which it fails to recover (Nimmo et al., 2015). In the context of forests relevant perturbations may relate to cutting of trees by humans, fire, drought or land-use change. Theory relating to ecosystem resilience suggests that these perturbations may interact either amongst each other or with other drivers resulting in positive feedbacks driving the system into a different state (Scheffer et al., 2001). However, despite their importance, these feedbacks can be difficult to detect because drivers may operate over spatial and temporal scales that vary by many orders of magnitude (Reyer et al., 2015). Of particular concern from the perspective of forests are interactions between local disturbances such as fire, pests, drought or deforestation and climatic changes that impair regeneration of tree species (Reyer et al., 2015). For example, it is thought that logging and deforestation in tropical forests combined with drought and increased frequency of fires may lead to a shift to savannah-like vegetation structure (Barlow and Peres, 2008; Nepstad et al., 1999). Similarly, in large disturbances in Mediterranean forests can lead to reduced seedling recruitment and invasion by grasses and shrubs, which result in increased fire frequency and further suppression of tree cover (Acácio et al., 2007). However, a number of high profile studies that used satellite based observation of tree cover to infer alternative stable states in both boreal (Scheffer et al., 2012) and tropical systems (Hirota et al., 2011) have come under criticism (Hanan et al., 2014) since apparent discontinuities in tree cover may, in part, be as a result of the methods used to produce global remote sensing products.

Despite the interest in the potential for transitions of forests to treeless states, there has been relatively little empirical research identifying feedback mechanisms that may cause these shifts. In addition the feedbacks that have been characterised are from regions where perturbations promote fire because of climatic interactions, even though dieback is also occurring in temperate regions that do not exhibit these feedbacks (Martin et al., 2015). Given that tree mortality as a result of climate change, pathogens and insect pests is increasing in many temperate forests (Seidl et al., 2014; van Mantgem et al., 2009) investigation of feedbacks that may drive these systems into non-forest states is needed to aid risk assessment.

Here we use a long-term data set, collected in a temperate forest ecosystem that has undergone partial stand dieback in recent decades. At this site in southern England, monitoring data have been collected repeatedly over a period of 50 years from 1964-2014 (Martin et al., 2015; Mountford and Peterken, 2003; Mountford et al., 1999). Using statistical models we investigate correlates of tree mortality and recruitment, both of which include potential feedback mechanisms. We then use results from these models and information from the literature to produce an individual based model to test the effects of identified feedbacks on forest structure. Specifically, using these methods our aims are:

1. To test the factors that influenced seedling and sapling density
2. To test the influence of tree size, growth rate, proximity of dead trees and soil characteristics on mature tree mortality
3. Test the influence of feedbacks identified in statistical models on forest structure using an individual based model

**Methods (aim for 1500 words, currently 2120)**

**Site description**

For this study we used data collected over a 50 year period in Denny Wood, which is located in the New Forest National Park, in Southern England (Lat: 50o 51.5’ N and 1o 32.5’ W). The site is gently sloping (1-3 degrees), with clay rich brown earth soils. Woodland vegetation is dominated by old-growth beech (*Fagus sylvatica*) with frequent pedunculate oak (*Quercus* robur) and birch (*Betula pendula, B. pubescens*), and an understory primarily of holly (*Ilex aquifolium*). In open areas the ground vegetation is mostly comprised of *Agrostis*-dominated grassland or stands of bracken (*Pteridium aquilinum*). There are large populations of deer, ponies and cattle in the New Forest, resulting in high herbivore pressure. The site which we used for this study is likely to have experienced high browsing pressure since the 1960’s (Mountford and Peterken, 2003). More detail on the site is given in Mountford *et al* (1999), Mountford and Peterken (2003) and Martin *et al* (2015).

**Data collection**

Measurements were conducted in one 20 m-wide 1 km long transect, which was originally established in the 1950s. The transect was subdivided into contiguous 45 20 x 20 m (0.04 ha) subplots and surveyed in 1964, 1984, 1988, 1996 and 2014. Details of measurements are presented Mountford *et al* (1999), Mountford and Peterken (2003) and Martin *et al* (2015). In each survey, the location and species name of all woody stems >1.3 m in height were recorded, their diameter at breast height measured using diameter tapes, and their status assessed as either alive or dead. Each stem >1.3m height was given a unique ID number to allow individual trees to be tracked. Stems <10 cm DBH were classified as saplings and those >10 cm DBH as mature.

In 2014 we collected data on seedling density, canopy openness, and soil characteristics. The density of tree seedlings of all species present in 10 x 10 m plots located in the centre of the 20 x 20 m plots was recorded. Canopy openness of subplots was assessed using a concave spherical densitometer in all four corners and the centre of 20 x 20 m plots, and the mean calculated for each subplot. Soil type was assessed by collecting 3 soil samples from within each 20 x 20m subplot using a 5 cm diameter soil corer. The first 20 cm of the mineral layer was retained. Soil samples were sent to the Forest Research laboratories in Surrey, UK where particle size distribution of the soil was determined by suspending 30 g of soil in water which was passed through the flow cell of a laser diffraction particle size analyser (Beckman Coulter LS13320).

**Statistical analysis**

**Recruitment of juveniles**

Our assessment of tree recruitment focussed on saplings (woody stems >1.3 m height and <10 cm DBH) and seedlings. We tested for relationships seedling density and canopy openness and grazing pressure using generalised linear models. We also tested the the effect of the density and basal area of mature trees (stems >10 cm DBH) on sapling density using generalised mixed models with a poisson error structure for the three major woody species present: beech, holly and oak. These models included interactions with survey year to identify if the slope of the relationship between mature tree density and sapling density changed over time. To determine whether the widespread loss of smaller stems seen in a previous study at the site (Martin *et al.* 2015) was largely attributable to growth of individuals into other size classes or mortality we tracked the fate of each sapling recorded. As part of this we also calculated the annual mortality rate, , as defined by Sheil, Burslem & Alder (1995):

where and are the number of stems at the first and second surveys respectively and is the number of years between censuses.

**Individual tree mortality**

We assessed whether some of the mature tree mortality observed in Denny wood could be explained by the self-thinning process using a linear mixed model with subplot number as a random effect to relate stem density to basal area. A negative slope suggests a gain in BA with a loss of stem density while a positive slope suggests a gain in BA with increasing stem density.

During data processing tree status (alive or dead) was coded at census time. Only trees with diameter measurements at both and prior to time were included in analyses. Censuses on the transect were undertaken 5 times from 1964-2014 with a mean (± SD) census interval period of 12.5 ± 6.7 years (range 4-20 years). Since trees used in mortality models required censuses at both and prior to time we selected three non-overlapping census periods: 1984-1988, 1988-1996 and 1996-2014 (mean census period 10 ± 5.9 years). Statistical models of individual tree mortality were developed using logistic mixed effects models, which describe the probability that a tree dies in a given period of time. To correct for the variation in census interval we used a complementary log-log link with an offset equal to the census interval so that predictions from models were equivalent to the annual probability of mortality (Fortin *et al.* 2008). Subplot ID number was used as a random effect to account for repeated sampling of the same plots (Fortin *et al.* 2008).

Models were developed in a four-step process similar to the workflow of Chao et al. (2008). In step 1 we prepared predictors classified into 4 groups:

* 1. tree size – DBH (cm), Basal area (m2), and tree size relative to other trees in the transect (bounded between 0 and 1)
  2. tree growth - Annual diameter growth rate (mm year-1), basal area growth rate (mm2 year-1), relative DBH growth rate (% DBH increase year-1), and relative BA growth rate (% BA increase year-1)
  3. proximity to dead trees - distance of an individual tree to a dead tree and abundance of dead trees in a 10m buffer
  4. soil type - percentage of each soil sample classified as sand

For more detail on the calculation of these variables see the supplementary materials. All model variables were standardised using the methods of Schielzeth (2010) by subtracting the mean of the variable and dividing by its standard deviation. This allows coefficients to be interpreted as effect sizes, reduces collinearity between variables and improves model convergence (Schielzeth, 2010).

In step 2 we selected the best predictor for each group by choosing the univariate logistic mixed effect models which had the lowest AICc (Burnham and Anderson, 2002; Chao et al., 2008a). This step reduces intercorrelation of variables which can lead to difficulty in interpreting effects (Chao et al., 2008b). In step 3 a full multivariate model was developed using these selected variables using additive terms only. In step 4 model averaging was used to produce parameter estimates for models with a ΔAICc≤7.

**Individual based model**

We used an individual based model, built using Netlogo (Wilensky, 1999) to investigate the importance of feedbacks in causing collapse of forest structure. The model description follows the ODD protocol for describing individual based models (Grimm et al., 2006).

**Purpose**

We used the model to investigate under what conditions loss of tree cover and basal area (BA) might occur in a simplified representation of a New Forest beech woodland. The only species represented is beech, as this is the dominant species found in the area, and mortality of the species has caused the majority of BA loss from 1964-2014 (Martin et al., 2015).

**Entities, state variables and scales**

The model comprises of two types of entities: grid cells and individuals. Individuals represent beech trees. Each individual is characterised by its location, development stage (juvenile or mature), age (in years), DBH (cm), basal area (m2), mean seed dispersal distance (mean distance from the source, in number of cells), growth rate in previous year (mm year-1), local basal area and stem density of mature trees (within a circles with an area of 400 m2, the size of plots we used for surveys), and distance to nearest dead tree (m) . DBH of mature trees and juveniles is derived from the age of trees using an equation for beech growth defined in Holzwarth *et al.* (2013) and BA defined as . Local basal area and stem density represent the total BA and stem density within a circle with an area of 400 m2 – equivalent to the plot size used in data collection. Dispersal distance is a random number drawn from an exponential distribution with a mean of 5 m.

All grid cells in the model are considered suitable for individuals. Each grid cell is characterised by its location, whether a tree has died in that patch and the time since last tree death on that patch. When a tree dies the patch value changes from 0 to 1 and after 10 ticks if no other tree has died on this cell this value returns to 0. Each grid cell can only contain one mature individual, but may contain multiple juveniles. The model landscape consists of 100 x 100 grid cells, with each cell representing 1 m2, thus the entire area represents 1 ha. Each model time step represents one year.

**Process overview and scheduling**

Initially the distribution of individuals is determined by randomly distributing 280 mature individuals with a random age drawn from an exponential distribution with a mean of 80 years assigned to each individual. This was approximately the density and age structure of Denny Wood when first surveyed in 1964. At the same time 100 juvenile trees are randomly distributed across the space. Then in each time step the following events are processed in the given order: increase age of individuals by one year, increase individual DBH & BA, identification of whether the time step represents a mast year, seed dispersal from mature trees > 40 years old, and death.

**Design concepts**

The total number of mature trees and basal area *emerge* from changes in the probability of mature tree mortality that occur as they age and increase in size, as well as from changes in the mortality of juvenile trees. *Interactions* between individuals are the result of density dependant mortality processes, which show size asymmetry. For juveniles this is modelled by defining a maximum number of juveniles that can coexist in the area as 1000, and a maximum number of juveniles per cell as 100. When these numbers are exceeded the smallest juveniles are killed. Similarly, for mature trees the local maximum BA was set at 75 m2 ha-1, the maximum observed for any plot during 1964-2014. When this maximum is exceeded the smallest mature tree in an area of 400 m2 is killed.

In addition our model tests the impact of reduced survival of juveniles in areas with few mature trees, as seen at our site, by allowing the user to switch this on and off. When turned on this causes juveniles in locations where mature tree density is <5 in the surrounding 400 m2 to have a 0.02 annual chance of survival, based on our observations that no saplings (trees >1.3 m tall, <10 cm DBH) were found in areas with tree density lower than this. Additionally, our model tests what impact an increase in the probability of mature mortality as a result of being close to a dead tree would have on forest structure. This can be switched on and off. When switched on this causes the annual mortality to be increased as a function of distance to nearest cell with a dead tree. The parameter estimate for this was derived from our statistical model of mature tree mortality described above.

*Stochasticity* is used in the model to define whether a given time step represents a mast year. Beech trees produce large amounts of seed once every 2-3 years in the UK (Packham et al., 2012) and thus we set a probability of 0.3 of each year being a mast year.

To test the model we first initiated the model so that forest structure was similar to that seen in 1964. We then ran different scenarios with and without differential juvenile mortality and spatial mortality feedbacks and compared the change in BA and forest structure after 50 time steps to those we observed in 2014. Following this we ran the models using the same parameterisation for a further 100 years to investigate possible future changes in BA and tree cover under the different scenarios. Each model run had 100 iterations and median values were used to summarise model results.

**Results (aim for ~600 words) – currently 539**

**Tree recruitment**

Mean beech seedling density (± SE) in 2014 was 115.22 ± 32.14 seedlings ha-1. Canopy openness was positively related to beech seedling density (slope=0.56 ± 0.09, P value <0.001), but no other variables were included in models which had ΔAICc≤7 and thus were considered to have poor support. No metrics of deer or pony density were related to seedling density in any way.

There was a negative relationship between sapling density and canopy openness. Only 6 plots had any saplings present, but all of these plots had a canopy openness of < 20%. There was a positive relationship between mature beech density and sapling density (slope= 1.27 ± 0.14, P value<0.0001), and sapling density tended to be reduced over time (slope= -0.69 ± 0.10, P value<0.0001). The relationship between mature beech density and sapling density grew more positive over time (interaction term=0.38 ± 0.09, P value<0.001), though since the density of mature trees was also reduced over this time period this effect was relatively unimportant in determining sapling densities. As the number of beech saplings declined during the years 1964-2014, so did the mortality rates of these saplings, from a maximum of 4.07% per year in 1964-1984 to 0.50% in 1996-2014 (Table 1). Conversely the proportion of saplings that became mature trees (>10 cm DBH) showed an increase over this time period (Table 1).

**Tree mortality**

The slope of the relationship between log subplot stem density and log subplot basal area was positive (slope=0.41 ± 0.05, marginal R2=0.24, Figure 1). However, in general subplots lost both stem density and basal area between 1964 and 2014 (Figure 2). Given that self-thinning processes tend to be strongest when plots are increasing in biomass and losing stem density at the same time (Coomes and Allen, 2007) such processes are unlikely to be responsible for the majority of tree death seen in Denny wood from 1964-2014.

When predicting the mortality of individual beech trees growth rate was considered the most important predictor, as it was included in all models with a ΔAICc≤7. Trees that grew slowly or shrunk were more likely to die than those that grew relatively quickly (slope=-0.59 ± 0.06, P value <0.001, Figure 3a). Next most important was tree DBH with an importance value of 0.8, and models suggested that tree size was positively correlated with probability of mortality (slope=0.21 ± 0.05, P value<0.001, Figure 3b). Distance to nearest dead tree and soil type were of similar importance with importance values of 0.52 and 0.45 respectively, with models indicating that trees closer to dead trees were more likely to subsequently die (slope=-0.24 ± 0.06, P value<0.001, Figure 3c) and trees located in areas of the forest where soils had higher sand content were less likely to die (slope=-0.27 ± 0.01, P value<0.001, Figure 3d).

**Individual based model**

Results from our individual based model suggest that when the annual probability of juvenile death is low, the forest does not undergo a transition to a treeless state even if feedbacks are present. However, when annual probability of juvenile death is >0.4 these feedbacks are enough to push the system into a decline, particularly when both distance to nearest dead tree causes an increase in the probability of mortality and juvenile survival is reduced in gaps (Figure 4). Thus the effect of feedbacks was dependent upon high background juvenile mortality.

**Discussion (aim for ~1200 words)**

1. Summary of results
2. Causes of death
3. Causes of seedling recruitment limitation
4. Feedbacks are important in our system, as seen in other transitions (~200 words)
5. This means that degradation of the forest could be self-perpetuating (~200 words)
6. Managers should attempt to break these feedback loops or stop them from being established (~200 words)

**Conclusion (200 words)**

1. Feedback loops can be important in causing transitions to non-forest states and need to be managed carefully (~200 words)

**References**

Acácio, V., Holmgren, M., Jansen, P.A., Schrotter, O., 2007. Multiple Recruitment Limitation Causes Arrested Succession in Mediterranean Cork Oak Systems. Ecosystems 10, 1220–1230. doi:10.1007/s10021-007-9089-9

Barlow, J., Peres, C.A., 2008. Fire-mediated dieback and compositional cascade in an Amazonian forest. Philos. Trans. R. Soc. B Biol. Sci. 363, 1787–1794. doi:10.1098/rstb.2007.0013

Burnham, K.P., Anderson, D.R., 2002. Model selection and multimodel inference: a practical information-theoretic approach, Ecological Modelling. doi:10.1016/j.ecolmodel.2003.11.004

Burrows, M.T., Schoeman, D.S., Buckley, L.B., Moore, P., Poloczanska, E.S., Brander, K.M., Brown, C., Bruno, J.F., Duarte, C.M., Halpern, B.S., Holding, J., Kappel, C. V., Kiessling, W., O’Connor, M.I., Pandolfi, J.M., Parmesan, C., Schwing, F.B., Sydeman, W.J., Richardson, a. J., 2011. The Pace of Shifting Climate in Marine and Terrestrial Ecosystems. Science (80-. ). 334, 652–655. doi:10.1126/science.1210288

Chao, K.J., Phillips, O.L., Gloor, E., Monteagudo, A., Torres-Lezama, A., Martínez, R.V., 2008a. Growth and wood density predict tree mortality in Amazon forests. J. Ecol. 96, 281–292. doi:10.1111/j.1365-2745.2007.01343.x

Chao, K.J., Phillips, O.L., Gloor, E., Monteagudo, A., Torres-Lezama, A., Martínez, R.V., 2008b. Growth and wood density predict tree mortality in Amazon forests. J. Ecol. 96, 281–292. doi:10.1111/j.1365-2745.2007.01343.x

Collet, C., Lantera, O., Pardos, M., 2001. Effects of canopy opening on height and diameter growth. Annu. For. Sci. 58, 127–134.

Coomes, D. a., Allen, R.B., 2007. Mortality and tree-size distributions in natural mixed-age forests. J. Ecol. 95, 27–40. doi:10.1111/j.1365-2745.2006.01179.x

Grimm, V., Berger, U., Bastiansen, F., Eliassen, S., Ginot, V., Giske, J., Goss-Custard, J., Grand, T., Heinz, S.K., Huse, G., Huth, A., Jepsen, J.U., Jørgensen, C., Mooij, W.M., Müller, B., Pe’er, G., Piou, C., Railsback, S.F., Robbins, A.M., Robbins, M.M., Rossmanith, E., Rüger, N., Strand, E., Souissi, S., Stillman, R.A., Vabø, R., Visser, U., DeAngelis, D.L., 2006. A standard protocol for describing individual-based and agent-based models. Ecol. Modell. 198, 115–126. doi:10.1016/j.ecolmodel.2006.04.023

Hanan, N.P., Tredennick, A.T., Prihodko, L., Bucini, G., Dohn, J., 2014. Analysis of stable states in global savannas: is the CART pulling the horse? Glob. Ecol. Biogeogr. 23, 259–263. doi:10.1111/geb.12122

Hirota, M., Holmgren, M., Van Nes, E.H., Scheffer, M., 2011. Global resilience of tropical forest and savanna to critical transitions. Science (80-. ). 334, 232–235. doi:10.1126/science.1210657

Holzwarth, F., Kahl, A., Bauhus, J., Wirth, C., 2013. Many ways to die - partitioning tree mortality dynamics in a near-natural mixed deciduous forest. J. Ecol. 101, 220–230. doi:10.1111/1365-2745.12015

Martin, P.A., Newton, A.C., Cantarello, E., Evans, P., 2015. Stand dieback and collapse in a temperate forest and its impact on forest structure and biodiversity. For. Ecol. Manage.

Mountford, E.P., Peterken, G.F., 2003. Long-term change and implications for the management of wood-pastures: experience over 40 years from Denny Wood, New Forest. Forestry 76, 19–43. doi:10.1093/forestry/76.1.19

Mountford, E.P., Peterken, G.F., Edwards, P.J., Manners, J.G., 1999. Long-term change in growth, mortality and regeneration of trees in Denny Wood, an old-growth wood-pasture in the New Forest (UK). Perspect. Plant Ecol. Evol. Syst. 2, 223–272. doi:10.1078/1433-8319-00072

Nepstad, D.C., Verissimo, A., Alencar, A., Nobre, C., Lima, E., Lefebvre, P., Schlesinger, P., Potter, C., Moutinho, P., Mendoza, E., Cochrane, M., Brooks, V., 1999. Large-scale impoverishment of Amazonian forests by logging and fire. Nature 398, 505–508. doi:10.1038/19066

Nimmo, D.G., Mac Nally, R., Cunningham, S.C., Haslem, a., Bennett, a. F., 2015. Vive la résistance: reviving resistance for 21st century conservation. Trends Ecol. Evol. 1–8. doi:10.1016/j.tree.2015.07.008

Olesen, C.R., Madsen, P., 2008. The impact of roe deer (Capreolus capreolus), seedbed, light and seed fall on natural beech (Fagus sylvatica) regeneration. For. Ecol. Manage. 255, 3962–3972. doi:10.1016/j.foreco.2008.03.050

Packham, J.R., Thomas, P. a., Atkinson, M.D., Degen, T., 2012. Biological Flora of the British Isles: Fagus sylvatica. J. Ecol. 100, 1557–1608. doi:10.1111/j.1365-2745.2012.02017.x

Reyer, C.P.O., Brouwers, N., Rammig, A., Brook, B.W., Epila, J., Grant, R.F., Holmgren, M., Langerwisch, F., Leuzinger, S., Lucht, W., Medlyn, B., Pfeifer, M., Steinkamp, J., Vanderwel, M.C., Verbeeck, H., Villela, D.M., 2015. Forest resilience and tipping points at different spatio-temporal scales: approaches and challenges. J. Ecol. 103, 5–15. doi:10.1111/1365-2745.12337

Scheffer, M., Carpenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413, 591–6. doi:10.1038/35098000

Scheffer, M., Hirota, M., Holmgren, M., Van Nes, E.H., Chapin, F.S., 2012. Thresholds for boreal biome transitions. Proc. Natl. Acad. Sci. U. S. A. 109, 21384–9. doi:10.1073/pnas.1219844110

Schielzeth, H., 2010. Simple means to improve the interpretability of regression coefficients. Methods Ecol. Evol. 1, 103–113. doi:10.1111/j.2041-210X.2010.00012.x

Scholes, R., Settele, J., Betts, R., Bunn, S., Leadley, P., Nepstad, D., Overpeck, J., Taboada, M.G., 2014. Terrestrial and inland water systems, in: Field, C., Barros, V., Mach, K., Mastrandrea, M. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Cambridge University Press, Cambridge, pp. 271–360.

Seidl, R., Schelhaas, M., Rammer, W., Verkerk, P.J., 2014. Increasing forest disturbances in Europe and their impact on carbon storage. Nat. Clim. Chang. 1–6. doi:10.1038/nclimate2318

Seidl, R., Spies, T. a., Peterson, D.L., Stephens, S.L., Hicke, J. a., 2015. Searching for resilience: addressing the impacts of changing disturbance regimes on forest ecosystem services. J. Appl. Ecol. n/a–n/a. doi:10.1111/1365-2664.12511

Sheil, D., Burslem, D.F.R.P., Alder, D., 1995. The Interpretation and misinterpretation of mortality rate measures. J. Ecol. 83, 331–333. doi:10.2307/2261571

Van Mantgem, P.J., Stephenson, N.L., Byrne, J.C., Daniels, L.D., Franklin, J.F., Fulé, P.Z., Harmon, M.E., Larson, A.J., Smith, J.M., Taylor, A.H., Veblen, T.T., 2009. Widespread increase of tree mortality rates in the western United States. Science 323, 521–524. doi:10.1126/science.1165000

Wilensky, U., 1999. Netlogo.

**Figures**



Figure 2 – Relationship between subplot stem density and total subplot basal area. Points represent individual plots in 1964 (red circles), 1996 (green triangles) and 2014 (blue squares). The solid line represents the prediction from a mixed model of this relationship with the grey band representing the coefficient confidence intervals. Note that that both x and y axes are log transformed.

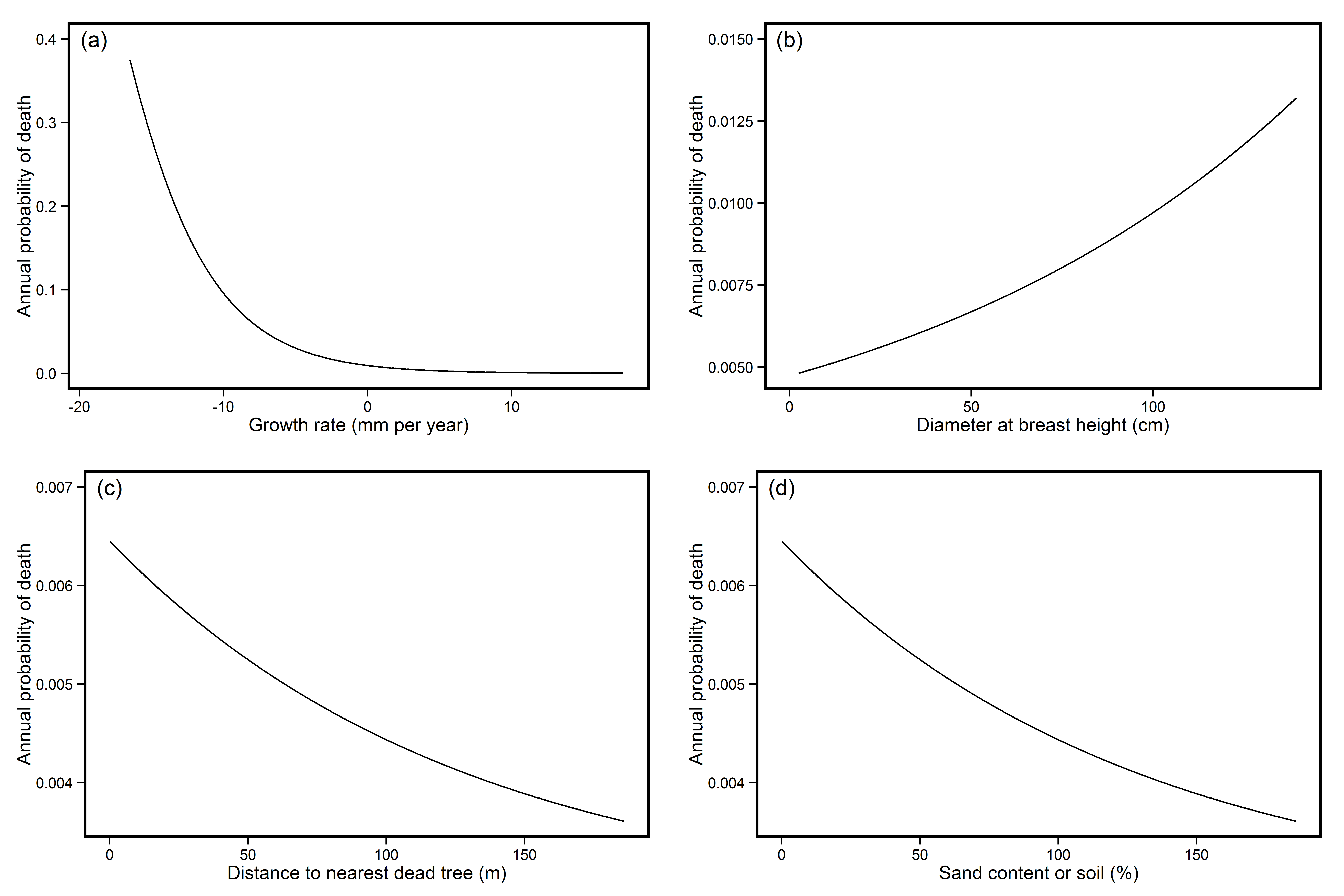


Figure 3 – Relationship between annual probability of beech tree death (a) growth rate per year, (b) diameter at breast height, (c) distance to nearest dead tree and (d) sand content of soil. Lines represent predictions generated from model-averaged parameter estimates.

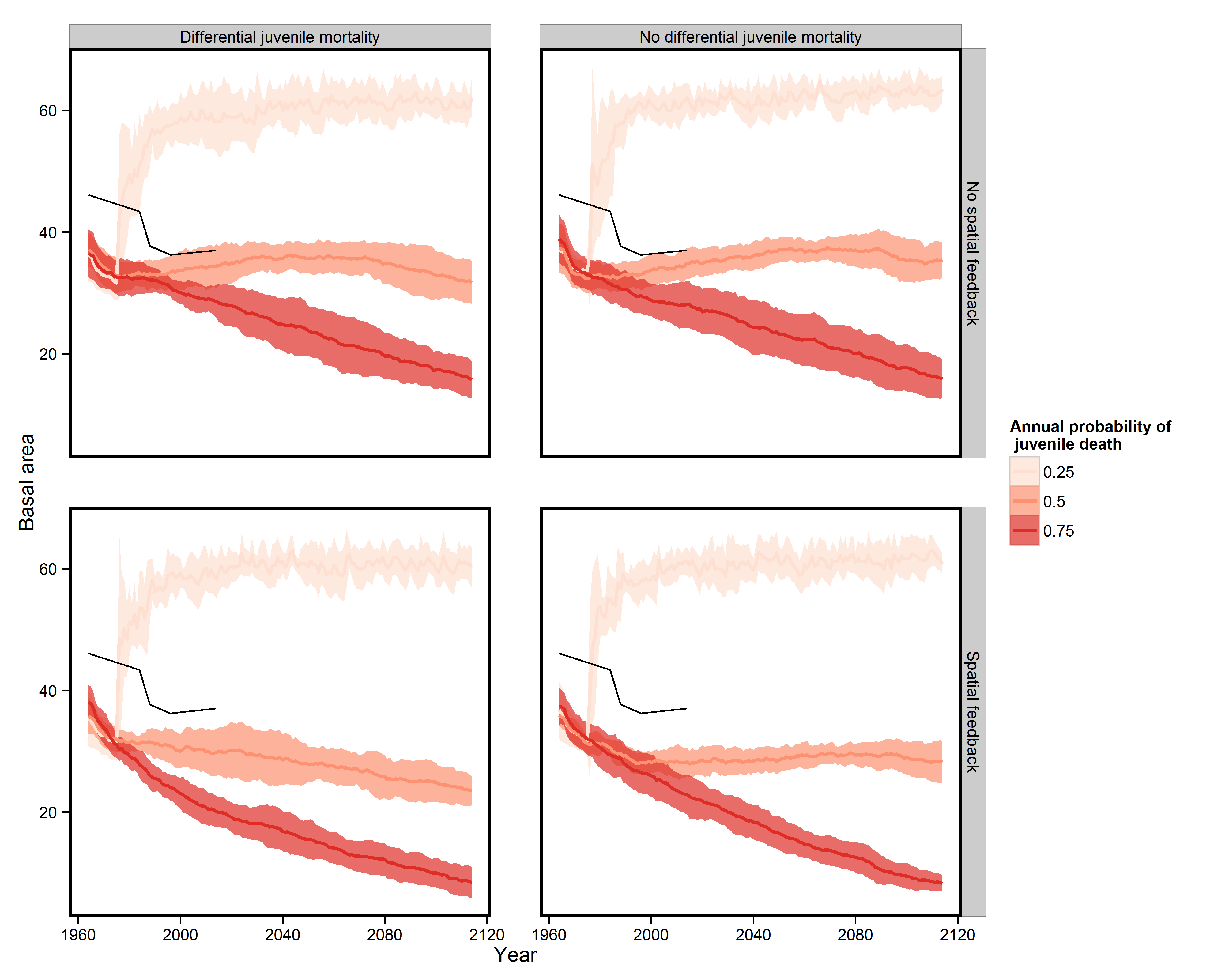


Figure 4 – The effect of feedbacks in mature tree death and juvenile mortality with varying probability of juvenile death on basal area from 1964 to 2114. Coloured lines represent mean basal area at each model time step with band representing the mean ± 1 SD. Black lines represent field observations from 1964 – 2014 to allow comparison between model results and real data.

Table 1 – Summary of recruitment and mortality of saplings in Denny wood from 1964 to 2014.

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Census period | Proportion of saplings that died | | | Proportion of saplings that increased to >10 cm DBH | | | Annual mortality rate of saplings | | |
|  | Beech | Oak | Holly | Beech | Oak | Holly | Beech | Oak | Holly |
| 1964-1984 | 0.56 | 0 | 0.78 | 0.14 | 1.00 | 0.04 | 4.07% | 0 | 7.20% |
| 1984-1988 | 0.11 | NA | 0.22 | 0.2 | NA | 0.01 | 2.79% | NA | 5.91% |
| 1988-1996 | 0.15 | NA | 0.25 | 0.32 | NA | 0.08 | 2.01% | NA | 3.53% |
| 1996-2014 | 0.09 | NA | 0.62 | 0.61 | NA | 0.16 | 0.50% | NA | 5.22% |

Table 2 – Coefficients of beech tree mortality from 1964 to 2014 produced from model averaging of mixed effect complementary log-log models with ΔAICc≤7.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Variable** | **Estimate** | **Std. Error** | **Lower CI** | **Upper CI** | **p value** | **Importance value** |
| Intercept | -5.12 | 0.16 | -5.43 | -4.82 | <0.0001 | 1 |
| DBH | 0.21 | 0.05 | 0.11 | 0.31 | <0.0001 | 0.8 |
| Distance to dead tree | -0.24 | 0.06 | -0.36 | -0.12 | <0.0001 | 0.53 |
| Growth rate | -0.59 | 0.06 | -0.70 | -0.47 | <0.0001 | 1 |
| Sand content | -0.27 | 0.01 | -0.29 | -0.26 | <0.0001 | 0.45 |

Table 3 – Parameter values for individual based model

|  |  |  |  |
| --- | --- | --- | --- |
| Parameter name | Sources | How derived | Value |
| Number of seedlings produced in mast year per tree | (Olesen and Madsen, 2008) | Mean of number of seedlings present after a mast year divided by the number of mature beech trees in the woodland. | 82 (26) |
| Juvenile height growth rate in gaps | (Collet et al., 2001) | Used values from reference | 9.5 cm year-1 |
| Juvenile height growth rate under closed canopy | (Collet et al., 2001) | Used values from reference | 1.2 cm year-1 |
| Maximum juvenile density | (Olesen and Madsen, 2008) | Used values from reference | 22 seedlings m-2 |
| Juvenile mortality in gaps | This study | Derived from statistical analyses |  |
|  |  |  |  |