Delayed effects of climate on vital rates leads to demographic divergence in Amazonian forest fragments

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Text of abstract. 34,281 year x individual measurements!

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# Introduction

Tropical deforestation driven by the expansion of agriculture and other human activities is a primary driver of biodiversity loss worldwide (Alroy, 2017; Haddad et al., 2015). Deforestation also results in landscapes where the remaining forest can be highly fragmented, with forest remnants of different sizes embedded in a matrix of often highly-contrasting habitat (Bianchi & Haig, 2013; Taubert et al., 2018). For example, over 24 million ha of the Brazilian Amazon have been cleared in the last two decades (Silva Junior et al., 2021), resulting in their extensive fragmentation and the creation of over 70,000 km of new forest edges annually (Broadbent et al., 2008). Fragmentation is associated with myriad ecological changes, including the local and regional extinction of plant species (da Silva & Tabarelli, 2000; Laurance et al., 2006). Although the demographic mechanisms responsible these extinctions are poorly understood (Bruna et al., 2009), it is often hypothesized that the dramatically altered environmental conditions in fragments (Arroyo-Rodríguez et al., 2017; Didham & Lawton, 1999; Ewers & Banks-Leite, 2013) are driving declines in reproduction, recruitment, or survivorship (Bruna, 1999; Laurance et al., 1998; Zartman et al., 2015). Despite the fact that this mechanism is thought to be particularly important in species-rich tropical forests (Betts et al., 2019; Didham & Lawton, 1999; Laurance et al., 2001), efforts to link population-level demographic responses with altered environmental conditions remains scarce.

Studies in temperate systems have shown that the demography of species can also be altered by climate change (Doak & Morris, 2010; Selwood et al., 2015; Sletvold, 2005; Williams et al., 2015), with more pronounced effects for fragmented populations (Holyoak & Heath, 2016; Oliver et al., 2015). While the demographic consequences for tropical species are expected to be similarly severe (Brodie et al., 2012; Scheffers et al., 2017), surprisingly little is known about the consequences of changing climates for taxa in tropical biomes, including Amazonia (Paniw et al., 2021). Climate models for the next 50-100 years predict that some parts of the Amazon basin will experience more frequent and severe droughts as well as more frequent periods of high precipitation (Cai et al., 2014; Duffy et al., 2015; Mora et al., 2013; Zeng et al., 2008). Plant populations may be particularly sensitive—an increase in the frequency and severity of extreme precipitation events can have detrimental consequences for survival and reproduction (Esteban et al., 2021; Gaoue et al., 2019). This increased sensitivity to climate fluctuations, coupled with the evidence that growth and survivorship are already lower in fragments (Bruna et al., 2002; Laurance et al., 1998; Zartman et al., 2015), has led to speculation that plants in tropical forest fragments are particularly at risk from drought and other climatic extremes (Laurance et al., 2001; Opdam & Wascher, 2004; Selwood et al., 2015).

Whether the demography of plant populations in tropical forest fragments is more susceptible to drought and other climatic extremes remains unclear for three primary reasons. First, most studies of plants in fragments have focused on a single life-history stage or process (Bruna et al., 2009; Ehrlen et al., 2016), making it challenging to draw broader demographic conclusions. Similarly, there is a growing literature on how tropical plants respond to droughts (Esquivel-Muelbert et al., 2019; González-M et al., 2020; Uriarte et al., 2016), but few studies have compared the responses of plants in continuous forest with those in forest fragments (Laurance et al., 2001). Finally, the multi-year data needed to test population-level hypotheses about climate change and fragmentation are scant, especially for tropical systems (Crone et al., 2011; Salguero-Gómez et al., 2015). These data are critical not simply because they allow for capturing variation in climatic conditions and the resulting demographic responses (Morris & Doak, 2002; Teller et al., 2016). They are also essential because while some demographic effects of fragmentation or drought can be detected immediately, others may take years to manifest (*e.g.,* Gagnon et al., 2011). Indeed, lagged responses of demographic vital rates to climate may in fact be the rule rather than the exception (Anderegg et al., 2015; Evers et al., 2021; Kannenberg et al., 2020; Schwalm et al., 2017).

Herbaceous plants represent up to 25% of plant diversity in tropical forests (Gentry & Dodson, 1987), are critical food and habitat for myriad species (Snow, 1981), and are economically and culturally vital (Nakazono et al., 2004; Ticktin, 2003). Nevertheless, the impacts of global change phenomena on their demography remain conspicuously understudied (Bruna et al., 2009). We used a decade of demographic and climatic data from an experimentally fragmented landscape in the Central Amazon to assess the effects of climate on populations of a tropical understory herb (*Heliconia acuminata*, Heliconiaceae). This time series, which included the severe droughts of 1997 (McPhaden, 1999) and 2005 (Marengo et al., 2008; Zeng et al., 2008), allowed us to address the following questions: (1) Does drought increase or decrease the growth, survival, and fertility of plant populations in continuous forest? (2) Are there delayed effects of drought on demographic vital rates, and if so what lag times are most critical? (3) Are the effects of drought on the vital rates of populations in fragments similar in direction and magnitude to those in continuous forest?

# Methods

## Study site

The Biological Dynamics of Forest Fragments Project (BDFFP) is located ~70 km north of Manaus, Brazil (2º30’ S, 60ºW). In addition to large areas of continuous forest, the BDFFP has forest fragment reserves isolated from 1980–1984 by felling the trees surrounding the area chosen for isolation and, in most cases, burning the downed trees once they dried (Bierregaard et al., 1992). In subsequent decades the vegetation regenerating around fragments has been periodically cleared to ensure fragment isolation (Bierregaard et al., 2001).

The BDFFP reserves are located in nonflooded (i.e., *terra firme*) tropical lowland forest with a 30–37m tall canopy (Rankin-de-Mérona et al., 1992) and an understory dominated by stemless palms (Scariot, 1999). The soils in the reserves are nutrient-poor xanthic ferrosols; their water retention capacity is poor despite having a high clay content. Mean annual temperature in the region is 26º C (range=19–39º C), and annual rainfall ranges from 1900–2300 mm. There is a pronounced dry season from June to October (Figure S1).

## Focal species

*Heliconia acuminata* (LC Rich.) (Heliconiaceae) is a perennial monocot distributed throughout Central Amazonia (Kress, 1990) and is the most abundant understory herb at the BDFFP (Ribeiro et al., 2010). While many *Heliconia* species grow in large patches in treefall gaps and other disturbed areas, others, such as *H. acuminata,* are found at lower densities in the darker and cooler forest understory (Rundel et al., 2020). These species produce fewer inflorescences and are pollinated by traplining rather than territorial hummingbirds (Bruna et al., 2004; Stouffer & Bierregaard, 1996). In our sites *H. acuminata* is pollinated by *Phaeothornis superciliosus* and *P. bourcieri* (Bruna et al., 2004). Plants begin flowering at the start of the rainy season; reproductive plants have flowering shoots (range = 1–7), each of which has an inflorescence with 20–25 flowers (Bruna & Kress, 2002). Fruits mature April-May, have 1–3 seeds per fruit (), and are eaten by a thrush and several species of manakin (Uriarte et al., 2011). Dispersed seeds germinate approximately 6 months after dispersal at the onset of the subsequent rainy season, with rates of germination and seedling establishment higher in continuous forest than forest fragments (Bruna, 1999; Bruna & Kress, 2002).

## Demographic data collection

In 1997–1998 we established a series of 5000 m2 plots (m) in the BDFFP reserves in which we marked and measured all *Heliconia acuminata*. The plots are located in continuous forest (), 10 ha fragments (), and 1-ha fragments (), with distance between plots ranging from 500 m–41 km. Plots in 1-ha fragments were on one randomly selected half of the fragment, plots in 10 ha fragments were in the fragment center, and plots in continuous forest were placed in locations 500–4000 m from the borders of secondary and mature forest. This study uses data only from 1-ha fragments and continuous forest sites because these represent extremes and because there are only three 10 ha fragment plots.

Each plot is subdivided into 50 quadrats (m) to simplify the annual surveys, during which we recorded the number of vegetative shoots each plant had, the height of each plant to the tallest leaf, and whether each plant was flowering (height and shoot number are correlated with leaf area, the probability of flowering, and rates of survivorship (Bruna, 2002; Bruna & Kress, 2002)). In this study, we used the product of shoot number and plant height as our measure of plant size. Preliminary analysis showed that the product of shoot number and height was a better predictor of total leaf area (which in turn is assumed to be a strong predictor of aboveground biomass) than either shoot number or height alone (Table S2 ). Plants that were not found for three consecutive surveys were considered dead. We also surveyed plots regularly during the rainy season to identify any that flowered after the survey. For additional details on the location of plots, survey methods, and *H. acuminata* population structure see Bruna & Kress (2002).

## Climate data

Data on precipitation and potential evapotranspiration in our sites were obtained from a published gridded dataset (0.25º 0.25º resolution) built using data from 3,625 ground-based weather stations across Brazil (Xavier et al., 2016). We used these data to calculate the standardized precipitation evapotranspiration index (SPEI) (Vicente-Serrano et al., 2010). SPEI is a proxy for meteorological drought that integrates precipitation and evapotranspiration anomalies over a specified time scale. Positive SPEI values for a given month indicate conditions wetter than the historical average for that month, while negative values of SPEI indicate droughts with intensity categorized as mild (0 to -1), moderate (-1 to -1.5), severe (-1.5 to -2), or extreme (< -2) (McKee et al., 1993). SPEI can be calculated to represent different temporal scales of drought; we used 3-month SPEI because—given its shallow roots and rhizome—*H. acuminata* relies primarily on soil moisture rather than deeper water sources that can persist for longer timescales (Vicente-Serrano et al., 2010). Note that 3-month SPEI is still monthly data—each month’s SPEI value simply takes into account precipitation and evapotranspiration of the previous three months. SPEI calculations were made using the SPEI package(Beguería & Vicente-Serrano, 2017). The timing of drought events based on these SPEI calculations is consistent with that resulting from SPEI calculated with other data sources, though the magnitude of drought sometimes differed (Figure S2; Table S1 ).

## Statistical Modeling of Vital Rates

To assess the effects of drought history on plant vital rates we used Distributed Lag Non-linear Models (DLNMs) (Gasparrini et al., 2017). DLNMs capture how potentially delayed effects of predictor variables (e.g. SPEI) affect an outcome (e.g. growth) well beyond the event period. They do so by fitting a bi-dimensional predictor-lag-response association spline, referred to as a crossbasis function. This models a non-linear relationship between predictor and response (e.g. between SPEI and vital rates) and allows the shape of that relationship to vary smoothly over lag time. Using the dlnm package (Gasparrini, 2011; R Core Team, 2020), we created crossbasis functions with possible lags from 0–36 months. We chose 36 months as a maximum lag because prior transplant experiments with *H. acuminata* showed they typically recover from transplant shock in less than 36 months (Bruna et al., 2002) so this is a reasonable upper bound for lagged effects of drought.

The crossbasis function was fit to the data in the context of a generalized additive model (GAM) with restricted maximum likelihood using the mgcv package (Wood, 2017). The general form of the vital rate () models was as follows:

where is a smooth function of plant size (natural log of height shoot number), fit using a penalized cubic regression spline, is the crossbasis function in which is the SPEI value during the census month of an observation (February) and is the SPEI months prior (see Gasparrini et al. 2017 for details). The crossbasis function, can also be written:

where the crossbasis function, , is composed of two marginal basis functions: the standard predictor-response function , and the additional lag-response function . These marginal functions are combined as a tensor product smooth such that the shape of one marginal function varies smoothly along the other dimension (see chapter 5 of Wood (2017) and Gasparrini et al. (2017) for more detail). Penalized cubic regression splines were used for both marginal bases of the crossbasis function, with 35 knots for the lag dimension (i.e. number of lagged SPEI values for each observation with 36 months as a maximum lag) and 3 knots for the drought response dimension to restrict the shape of the fitted response to drought to bimodal when most complex. Because of penalization, the number of knots is generally not important as long it is large enough to allow the smooth to represent the ‘true’ relationship (Wood, 2017). Estimated degrees of freedom (edf) represent the ‘true’ complexity of the smooth after penalization with edf = 1 being equivalent to a straight line and larger numbers representing more complex curves.

To determine if plot characteristics influenced average vital rates we included a random effect of plot ID on the intercept; this was represented by in eq. 1. We determined the effects of SPEI on plant growth using plant size in year t+1 as a response variable. This was modeled with a scaled t family error distribution because residuals were leptokurtic with a Gaussian error structure. Because number of inflorescences was highly zero-inflated, we converted this to a binary response to model reproduction (i.e., 1 for ≥1 inflorescence, 0 for no inflorescences). We modeled both reproduction and survival (i.e., from year t to year t+1) using a binomial family error distribution with a logit link function. We modeled a potential cost of reproduction by including flowering in the previous year as covariate, , in eq. 1.

In the process of fitting the models, the penalty on the crossbasis smooth (and other smoothed terms) is optimized such that more linear shapes are favored unless the data supports non-linearity (Wood, 2017). We applied an additional penalty to shrink linear portions toward zero with the select=TRUE option to the gam() function, and inferred statistical significance of model terms with p-values from the summary.gam() function as recommended in Marra & Wood (2011).

The dlnm package does not currently allow modeling of factor by smooth interactions. This means we could not include habitat as an interaction term. We therefore fit separate models for plants in fragments and in continuous forest to allow the shape of the crossbasis function to differ between habitats.

All analyses were conducted in R version 4.0.2 (2020-06-22) (R Core Team, 2020).

# Results

Meteorological droughts in our focal region, as indicated by SPEI, are generally consistent with those reported in the literature (Table S1). For example, the drought associated with the 1997 El Niño Southern Oscillation (ENSO) event was one of the most severe on record for the Amazon (Williamson et al., 2000); correspondingly, 1997 has the lowest SPEI values in our timeseries (Figure 1d). The 2005 dry season (June–October) was also reported as an exceptionally dry year, although this drought mostly affected the southwestern Amazon (Marengo et al., 2008; Zeng et al., 2008). Our SPEI data show the 2005 dry season to be a moderate drought (-1 > SPEI > -1.5).

## Survival, growth, and flowering in continuous forest vs. fragments

Our dataset comprised 4,083 plants in continuous forest and 1,010 plants in forest fragments. Plots in CF had on average 2.7-fold more plants than plots in 1-ha fragments (CF = 681 ± 493 SD; 1-ha = 253 ± 30 SD).

When summarizing across years and plots, the proportion of *Heliconia acuminata* that survived in CF and 1-ha was similarly high (; Figure 1b). The proportion of surviving plants was lowest in the 2003–2004 transition year (), which coincided with droughts in both the 2003 and 2004 rainy seasons (Figure 1b,d) and was preceded by a drop in average plant size in the 2002–2003 transition year (Figure 1a). The lowest survival for 1-ha fragment plots () was for the 2005–2006 transition year, which encompassed a moderate drought in October 2005 and and wetter than average conditions (SPEI > 0.5) in December 2005 and January 2006 (Figure 1b,d). Survival was size dependent in both continuous forest and 1-ha fragments ( for the effect of log-transformed plant size in year t on survival in year t+1 in both habitats). While the survival probability of large plants approached 1 in both habitat types (Figure 2b), the survival of the smallest plants was higher in 1-ha fragments. However, the 95% confidence intervals for 1-ha fragments and continuous forest overlapped for all sizes.

Plants in continuous forest had an average of 2.9 shoots (± 1.8 SD) and were on average 40.6 cm tall (± 26.5 SD). Plants in 1-ha fragments had on average ~13.8% fewer shoots (2.5 ± 1.5 SD) and were ~10.8% shorter (36.3 cm ± 24.1 SD). Because the proxy for plant size used in our models was the product of these two metrics, plants in continuous forest were on average substantially larger than those in forest fragments (150 ± 175 SD vs. 112 ± 141 SD, respectively). The distribution of sizes of individuals was somewhat bimodal in both habitats and the major difference in plant size was in the larger of the two modes being smaller in fragments (Figure 2d). This disparity in plant size was most pronounced in the initial years of our surveys and then diminished over time (Figure 1a). Mean plant size dropped dramatically in 2003 in both habitat types, corresponding with a severe drought during the February census (SPEI = -1.39) (Figure 1d). As with survival, size in year t was a significant predictor of size in year t+1 ( in both habitats). While the effect was generally similar across size classes and habitat types, the impact of plant size on growth was greatest for mid-sized plants in continuous forest (Figure 2a).

While the overall proportion of all plants flowering was very low, it was nevertheless almost 40% higher in continuous forest than 1-ha fragments (0.05 ± 0.21 vs. 0.04 ± 0.19, respectively). This disparity was largely due to the fact that flowering is size-dependent ( in both habitats), with the probability of flowering increasing dramatically once plants reached the threshold size of about 148 (log(size) > 5 in Figure 2c). Even though flowering probability of large plants was greater in 1-ha fragments compared to continuous forest, there were proportionally far fewer plants above the reproductive size threshold in forest fragments (Figure 2d). Indeed, the most striking difference between habitat types coincided with a severe drought in 2003, when the percentage of flowering reproductive-sized plants was 28% in continuous forest vs. only 13.6% in 1-ha fragments (Figure 1c).

## Delayed effects of drought on demographic vital rates

Drought history had a significant () effect on the survival, growth, and flowering of plants in both habitats. Comparing the respective crossbasis surfaces, however, reveals that the specific climatic drivers, their timing, and their impact on individual vital rates all differed among habitats.

For 1-ha fragments, SPEI in the preceding 12 months had the strongest effect on survival, with the highest survival near SPEI of 0 and mortality increasing as conditions became either drier or wetter (i.e., as SPEI values became increasingly negative or positive, respectively; Figure 3a). In contrast, the effect of recent SPEI in continuous forest was both weaker and unidirectional—survival probability was reduced only when SPEI was < -1 (i.e., mild-severe drought), and even then only slightly (Figure 3b). Instead the most pronounced effects on the survival of plants in continuous forest were of SPEI at a lag time of 15–20 months (i.e. two dry seasons prior to a census) and 32–36 months. Drought (i.e., SPEI < -1) 15–20 months prior to a census was associated with reduced survival, while high precipitation (i.e., SPEI > 1) was associated with higher survival (put another way, there was a nearly linear relationship between SPEI and survival probability). Finally, plants in both habitat types showed an increase in survival probability with very high SPEI values (i.e., extremely high precipitation) at a lag time of 36 months. It should be noted, however, that only the first year of census data (1999) met these conditions. We compared the effects of SPEI history in continuous forest and fragments by subtracting the fitted values in Figure 3b from Figure 3a to produce Figure 3c. This shows that in average conditions (SPEI = 0), there is little difference in survival probability between continuous forest and forest fragments (Figure 3c). However, under extreme conditions, survival probability is higher in continuous forest by up to 0.025.

While the fitted crossbasis functions for fragments and continuous forest showed generally similar patterns of drought effects on growth (i.e, trends in plant size), the crossbasis function for 1-ha fragments indicated more complex responses in some situations (edf = 17.9 for 1-ha fragments; edf = 12.9 for continuous forest; see also Figure 4). For example, when SPEI = 0 (i.e., average conditions), growth is similar or slightly higher in continuous forest over all lag periods (Figure 4c). However, when the current wet season is unusually wet (SPEI>2), plants in continuous forests were larger by up to log(size) = 0.57 due to reduced growth in fragments. In contrast, drought at lags of 8–11 months (i.e., the end of the preceding year’s wet season) led to increased growth in both habitats, but the magnitude of the response was greater in 1-ha fragments.

Overall, the the probability of flowering was higher in continuous forest than in 1-ha fragments for all values of SPEI (Figure 5). The responses in 1-ha fragments were also more muted (1-ha edf = 8.1, continuous forest edf = 10.3), as indicated by the shape of the crossbasis function (Figure 5a). This led to some important inter-habitat differences in plant responses to prior droughts. Recent drought (i.e., at lag = 0 with SPEI < -1) increased the probability of flowering in continuous forest, as did drought at lags 15–20 (i.e., two dry seasons prior). However, drought at lags 7–13 (the end of the rainy season one year prior) reduced flowering probability in continuous forest far more than it did in fragments (Figure 5c).

Finally, with the exception of survival in 1-ha fragments (), the delayed effects of SPEI on all three vital rates varied significantly among plots ( for the random effect of plot). We found no evidence for a cost of reproduction. In both forest and fragments, flowering in the previous year was significantly positively related to growth (CF: , 1-ha: ) and flowering ( for both CF and 1-ha) meaning that plants which had flowered in the previous year were more likely to be larger and flower again, on average.

# Discussion

Understanding how landscape structure and abiotic conditions act to influence population dynamics is central to many conceptual frameworks for studying and conserving fragmented landscapes (Didham et al., 2012; Driscoll et al., 2013). Our results support the emerging consensus that the effects of climatic extremes on demographic vital rates can be delayed for months or even years (Evers et al., 2021; Teller et al., 2016; Tenhumberg et al., 2018). We also found that the delayed responses of populations in fragments can differ significantly in magnitude, direction, and lag time from those of populations in continuous forest. This suggests that the hypothesized synergies between climate and fragmentation on population dynamics (Laurance & Williamson, 2001; Opdam & Wascher, 2004; Selwood et al., 2015) are likely to be pervasive, but also far more complex than previously thought.

## Temporal variation in demographic responses to forest fragmentation

Many studies investigating the biological consequences of habitat fragmentation on plant growth, survival, and reproduction comprise short-term (<3 year) experiments and observations. Our results underscore the difficulty in extrapolating long-term trends from such short-term studies, particularly when studying long-lived organisms or when the responses of interest can vary with size or age. For instance, one would have reached a very different conclusion regarding the effect of fragmentation on annual survival if the study windows were 1999–2002 (i.e., higher survival in continuous forest), 2002–2005 (i.e., higher survival in fragments), or 2004–2007 (i.e, no clear effect of fragmentation) (Figure 1b). It is only when evaluating over longer time windows that it becomes apparent mortality is elevated in fragments relative to continuous forest (Figure ??), and that the observed interannual variation is largely driven by dynamic patterns of recruitment (Bruna, 2002) coupled with low mortality for plants beyond the smallest size classes (Bruna, 2003).

Similarly, conclusions regarding the effects of fragmentation on flowering—which is also both rare and size-dependent (Brooks et al., 2019)—would also differ based on the year in which they were investigated. This could lead to erroneous extrapolations regarding the effects of fragmentation on reproductive mutualists or population genetic structure (Côrtes et al., 2013; Uriarte et al., 2010; Uriarte et al., 2011). Conclusions based on short-term observations of temporally variable vital rates could lead to conservation and management practices that are ineffective or even counterproductive, especially when when failing to consider how the consequences of this variation might be modulated by organismal life history (Morris et al., 2008).

It is important to emphasize, however, that the overall effects of SPEI on survival and growth are more severe in fragments than continuous forest (Figures 3, 4). Furthermore, the magnitude of plant responses to climatic extremes is also greater in habitat fragments—extreme drought in dry seasons and extreme precipitation in during rainy seasons are most detrimental to growth and survival in fragments. While intact forest and its canopy buffer populations from climatic extremes, populations in fragments—especially near edges with high contrast matrix—likely lack this protection (Didham & Lawton, 1999; Ewers & Banks-Leite, 2013). We suggest it is these climate extremes, rather than trends in average temperature, precipitation, or SPEI (Laurance et al., 2014), that that are the causal mechanism underlying reduced plant growth and survival in forest fragments.

## Delayed effects of climate on demographic vital rates

Climate anomalies are known to have immediate effects on the growth, survival, or reproduction of plants (Esteban et al., 2021; Wright & Calderon, 2006), including *Heliconia* (Stiles, 1975; Westerband et al., 2017) and other tropical herbs (Wright, 1992). These effects can be complex or even contradictory—mild droughts can increase the growth rates of tropical trees and seedling survival, perhaps due to reductions in cloud cover and concomitant increases in solar radiation (Alfaro-Sánchez et al., 2017; Condit et al., 2004; Huete et al., 2006; Jones et al., 2014; Uriarte et al., 2018), but in severe drought years growth can be extremely low and mortality can be sharply elevated (Connell & Green, 2000; Edwards & Krockenberger, 2006; Engelbrecht et al., 2002). There is also evidence that the effects can persist for multiple years (Phillips et al., 2010), such as a boom in drought-year fruit production followed by severe post-drought “famine” (Pau et al., 2013; Wright et al., 1999).

Despite these insights, models of plant population dynamics rarely include the effects of environmental drivers [but see Williams et al. (2015); Tenhumberg et al. (2018); Molowny-Horas et al. (2017)). This has largely been due to the challenge (both ecologically and statistically) of detecting any demographic responses to climatic extremes that are delayed for multiple growing seasons. To address this, researchers have begun to use a number of statistical methods that test for time lags in demographic responses without *a priori* assumptions about the influence of any particular climate window (Evers et al., 2021; Teller et al., 2016; Tenhumberg et al., 2018). Our expansion of this approach, which offers an unbiased way of identifying these delayed effects without overfitting (but see Pierre et al. (2020) and Ogle et al. (2015) for alternative methods) yielded results consistent with this emerging literature—that the effects of precipitation extremes on the demography of *Heliconia acuminata* could be delayed for up to 3 growing seasons.

While it appears that delayed effects of climate on demographic vital rates may be ubiquitous (Evers et al., 2021), the extent to which they vary spatially or with habitat remains an open question. Our results clearly indicate that they can, with habitat-specific differences in how environmental conditions influenced future vital rates. For example, extreme values of SPEI—both positive (unusually high precipitation) and negative (drought conditions)—led to declines in the probability of individual survival in both habitat types. However, the magnitude of these declines was far greater in forest fragments. Similarly, the detrimental effects of extremes in SPEI on growth rates were also more pronounced in fragments. In contrast, variation in SPEI had far stronger effects on the probability of flowering in continuous forest than fragments. These results should be interpreted with some caution, however, as the relatively low number of plants in fragments that are above the threshold-size for flowering could limit the power to detect delayed effects.

There are several, non-mutually-exclusive explanations for delayed effects of SPEI on demography. The first is that the physiological processes underlying vital rates might be initiated long before they are demographically apparent (Evers et al., 2021), and hence be shaped by climatic events at any point in that physiological window. For example, the flowering shoots of *Heliconia chartacea* begin to develop 6–10 months prior to the appearance of inflorescences (Criley & Lekawatana, 1994). Adverse conditions during the 6 months following initiation, rather than the months when inflorescences are starting expand, leads to the aborted production of flowering shoots. Interestingly, we observed the opposite effect—drought conditions increased the probability of flowering two years later. While this could reflect bet-hedging in response to stress (Nihad et al., 2018), this does not appear to be the case, as growth or survival do not appear to decrease following reproduction (see also (Horvitz & Schemske, 1988). In fact, flowering in one year is associated with increased reproduction and growth in the next .

Demographic responses will also be delayed if abiotic stress causes plants to invest in belowground rhizomes (*sensu* Pumisutapon et al., 2012). The carbohydrates stored in rhizomes allow *Heliconia* to regenerate aboveground biomass following damage (Rundel et al., 1998) and protect the buds that give rise to new shoots from stressful conditions (Klimešová et al., 2018). This may be why drought led to delayed increases in growth—by shedding shoots and leaves (Bruna et al., 2002) and investing in rhizomes, plants are generating proportionately more buds with which to regenerate when conditions improve. This would also be consistent with the results of prior experiments, in which the growth rates of *H. acuminata* 8 months after they were mechanically damaged far exceeded those of control plants (Bruna & Ribeiro, 2005).

Third, it may be that the delayed demographic effects we observed are indirectly mediated by the effect of SPEI on other species rather than the direct effects on individual physiology (Evers et al., 2021). For example, topical trees may not die until three or more years after a drought (Criley & Lekawatana, 1994). When they finally do, the resulting leaf drop (Janssen et al., 2021) and treefalls allow for light penetration to the forest understory (Canham et al., 1990; Leitold et al., 2018), triggering a boom in the growth and flowering of understory plants (Bruna & Oli, 2005). Similar delayed changes in the local environment could also influence the foraging behavior of a plant’s pollinators (Bruna et al., 2004; Stouffer & Bierregaard, 1996), seed dispersers (Uriarte et al., 2011), or herbivores (Scott et al., 2021). While more work is needed to explain why the (delayed) effects of SPEI on *H. acuminata* survival and growth are greater in fragments than forest interiors, one hypothesis, motivated by recent intriguing results from other systems (Sapsford et al., 2017), is that the greater litterfall on edges (Vasconcelos & Luizão, 2004) may be altering the abundance of pathogens or mycorrhizae.

Finally, demographic delays could be an artifact of the timing of responses in relation to the census date. If extreme drought in the dry season before the census increased plant mortality during that season, for example, this would nevertheless appear in models as a delayed effect (e.g. in Figure 3b). In our case, this potential explanation for delayed effects applies only to plant size and survival, as plots were surveyed regularly throughout the reproductive season to identify flowering plants. This possibility is not unique to our study, rather it is a consequence of conducting demographic censuses on an annual scale while the climate is quantified monthly or seasonally. While the very slow growth and extremely low mortality rates of *H. acuminata* mean this effect is unlikely to be acting in our system, it may be that for some species it will be important to conduct demographic surveys at the same temporal scale at which climate is aggregated.

## Conclusions & Future Directions

Climate models for the Amazon predict a future of extremes—increases in the frequency and geographic extent of droughts, but also increases in the frequency and area affected by periods of unusual wetness (Duffy et al., 2015). Our results support the hypothesis that populations in fragments could be more susceptible to the effects of changing climate than those in continuous forest (Laurance et al., 2014). However, they also indicate that the demographic responses to climate change of populations in fragmented landscapes may be far more complex than previously appreciated. Multi-factorial, multi-season experiments (Aguirre et al., 2021; *sensu* Bruna & Ribeiro, 2005; Markewitz et al., 2010; Westerband et al., 2017), ideally manipulating multiple combinations of climatic variables (Mundim & Bruna, 2016), are needed to determine how and why habitat-specific differences in environmental conditions interact to delay the demographic responses of plants to climatic variability. Also needed are statistical tools that can test for synergistic effects of fragmentation and climate in vital rates, as those currently available do not allow for including interaction terms. This also limits the ability to include size by climate interactions in a DLNM; although plant responses to both fragmentation and climatic extremes can be size-specific (Bruna & Oli, 2005; Schwartz et al., 2019). The ability to identify size-specific lagged responses may be especially complicated given size and growth are rarely measured at the same time scale as SPEI and other putative climatic drivers.

Finally, no analytical approach assessing the potential for demographic lags can compensate for a lack of long-term data (Evers et al., 2021; Tenhumberg et al., 2018). Unfortunately, long-term data monitoring the entire life-cycle of tropical taxa are rare, and those doing so in fragmented landscapes are virtually nonexistent (Bruna & Ribeiro, 2005). Without investing in collecting such data, generalizations regarding the demographic consequences of climate change in these species rich and increasingly fragmented habitats will continue to prove elusive. More generally, however, researchers need to consider how delayed responses to climate could influence the interpretation of data in studies where the organisms lifespan exceeds the study’s duration.

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# Figures



Figure 1: Timeseries of demographic parameters and drought occurrence. Mean fold-change in plant plant size (log2(sizet+1 / sizet)) (a) shows that plant growth varied by year and habitat (1 ha fragments in orange, solid; continuous forest in blue, dashed). In most years plants grew on average with a notable exception in 2003 where plants regressed in size on average in both habitats (fold-change < 0). Error bars in a) represent standard deviation. The proportion of plants surviving each transition year (b) shows that average survival is high and was lowest in the 2004 census. The proportion of reproductive sized plants (c) is on average low and fluctuates substantially year to year. The cutoff for reproductive size plants in panel (c) is defined as the upper 90th percentile size of flowering plants in all years. Monthly 3-month SPEI is shown (d) with gray lines representing values from different grid cells encompassing BDFFP and the dark line representing the site mean. Yellow, orange, dark orange, and red stripes show mild, moderate, severe, and extreme drought, respectively.



(#fig:surv\_curve)Survivorship of plants labeled in the first survey year, 1998, which comprise 49% of the plants in the full dataset. After 10 years, 79.7% (1629/2055) of plants in continuous forest survived and 72.4% (393/543) of plants in fragments survived.

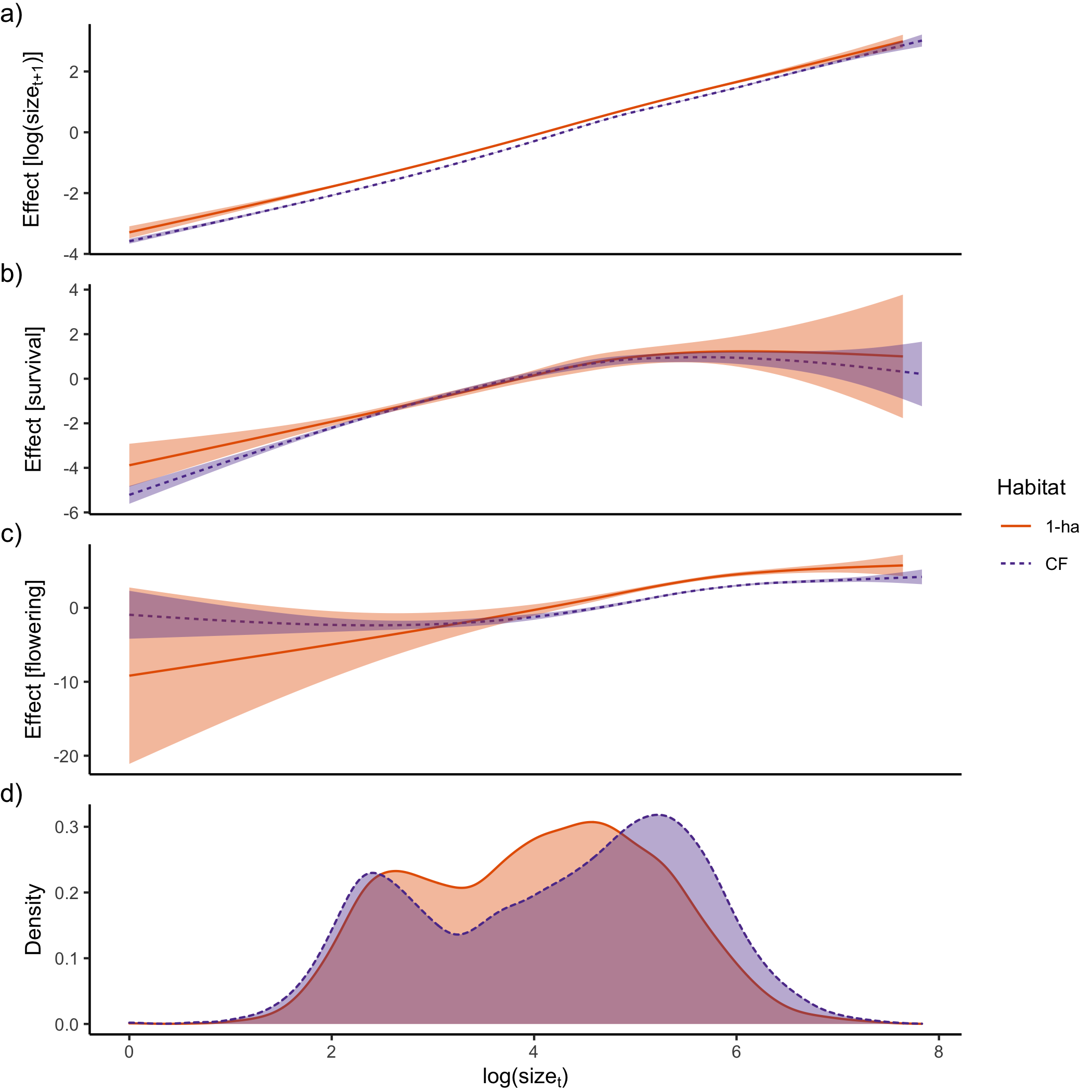


Figure 2: Smooth effect of plant size in the previous census from models for a) survival, b) log(size), and c) flowering probability, corresponding to the additive term s1(zi) in eq. 1. 95% confidence intervals are shown and include uncertainty in the intercept and uncertainty due to smoothness selection. The smooths for 1-ha fragments and continuous forest are fit in separate models. Panel d) shows the distribution of plant sizes in the two habitat types.

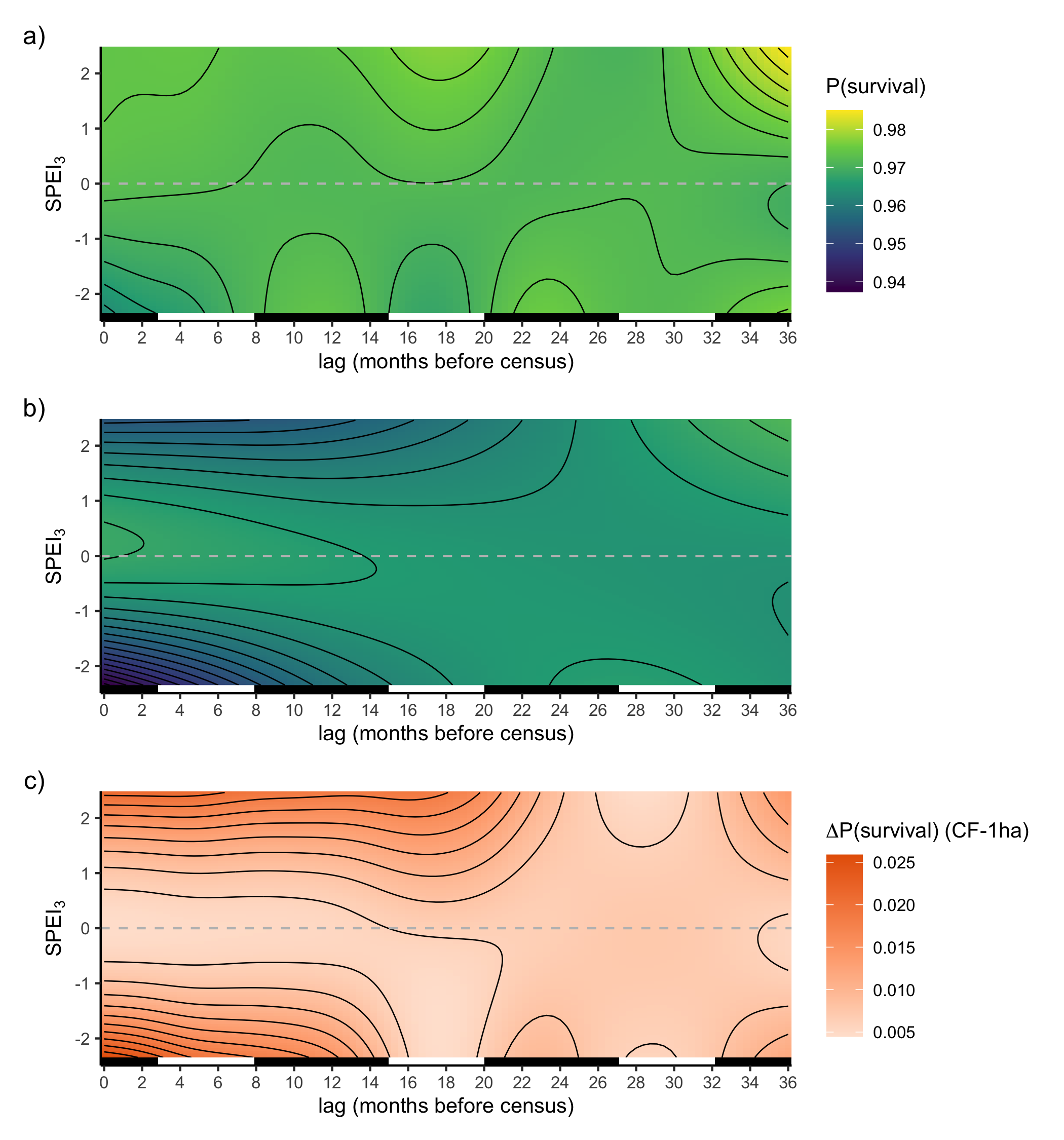


Figure 3: Smooth effect of lagged SPEI on survival in a) continuous forest and b) 1-ha fragments. Panel c) shows the difference between panels a) and b). Surface is modeled as a crossbasis function with cubic regression splines for each marginal basis. Model intercepts were added to fitted values of the crossbasis function and back-transformed to the response scale. Contour lines correspond to a change of 0.002. The bar on the bottom of each panel indicates wet seasons (black, November–May) and dry seasons (white, June–October).

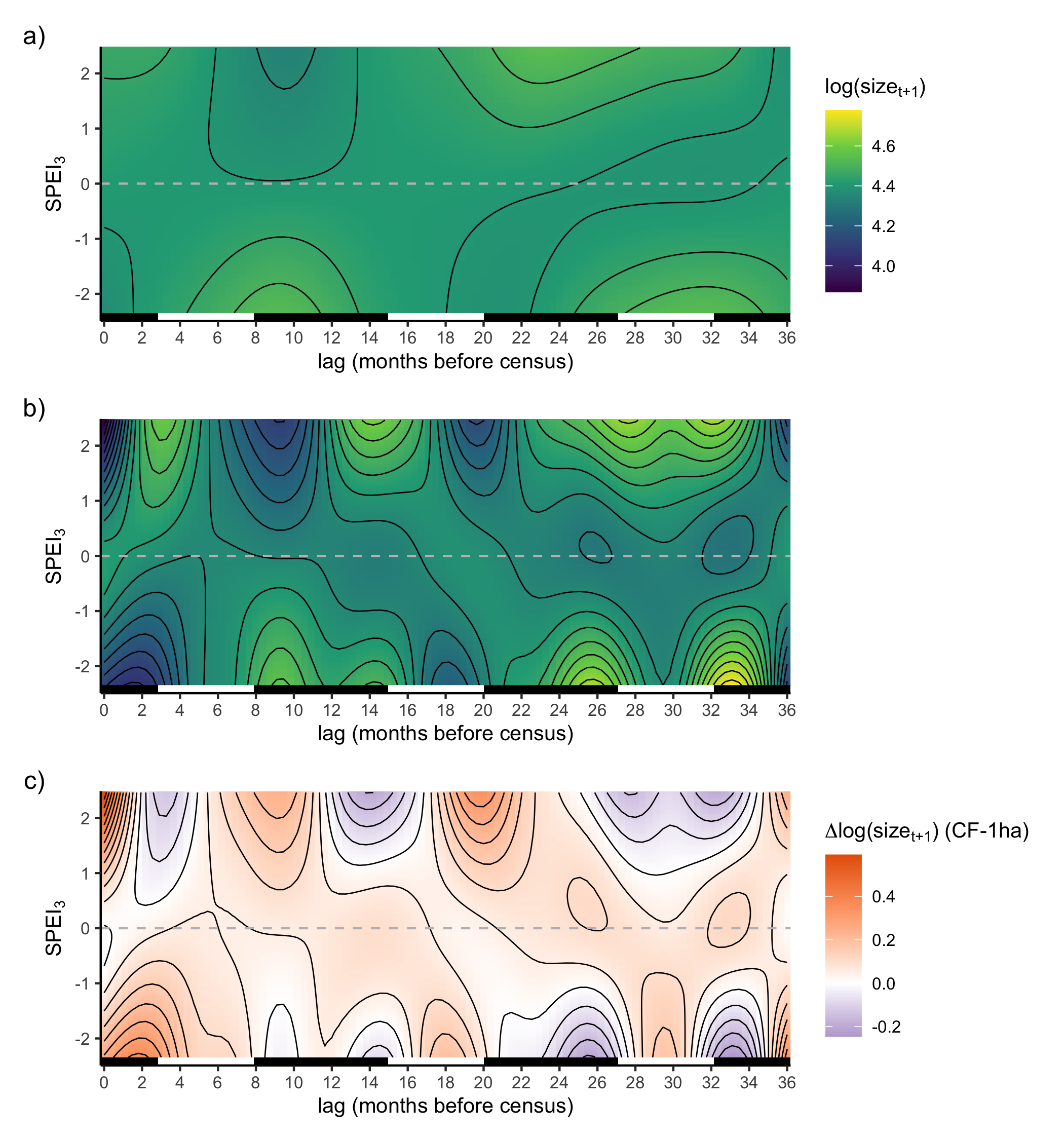


Figure 4: Smooth effect of lagged SPEI on plant growth for a) continuous forest and b) 1-ha fragments. Panel c) shows the difference between panels a) and b). Surface is modeled as a crossbasis function with cubic regression splines for each marginal basis. Model intercepts were added to fitted values of the crossbasis function. Contour lines correspond to a change of 0.05. The bar on the bottom of each panel indicates wet seasons (black, November–May) and dry seasons (white, June–October).

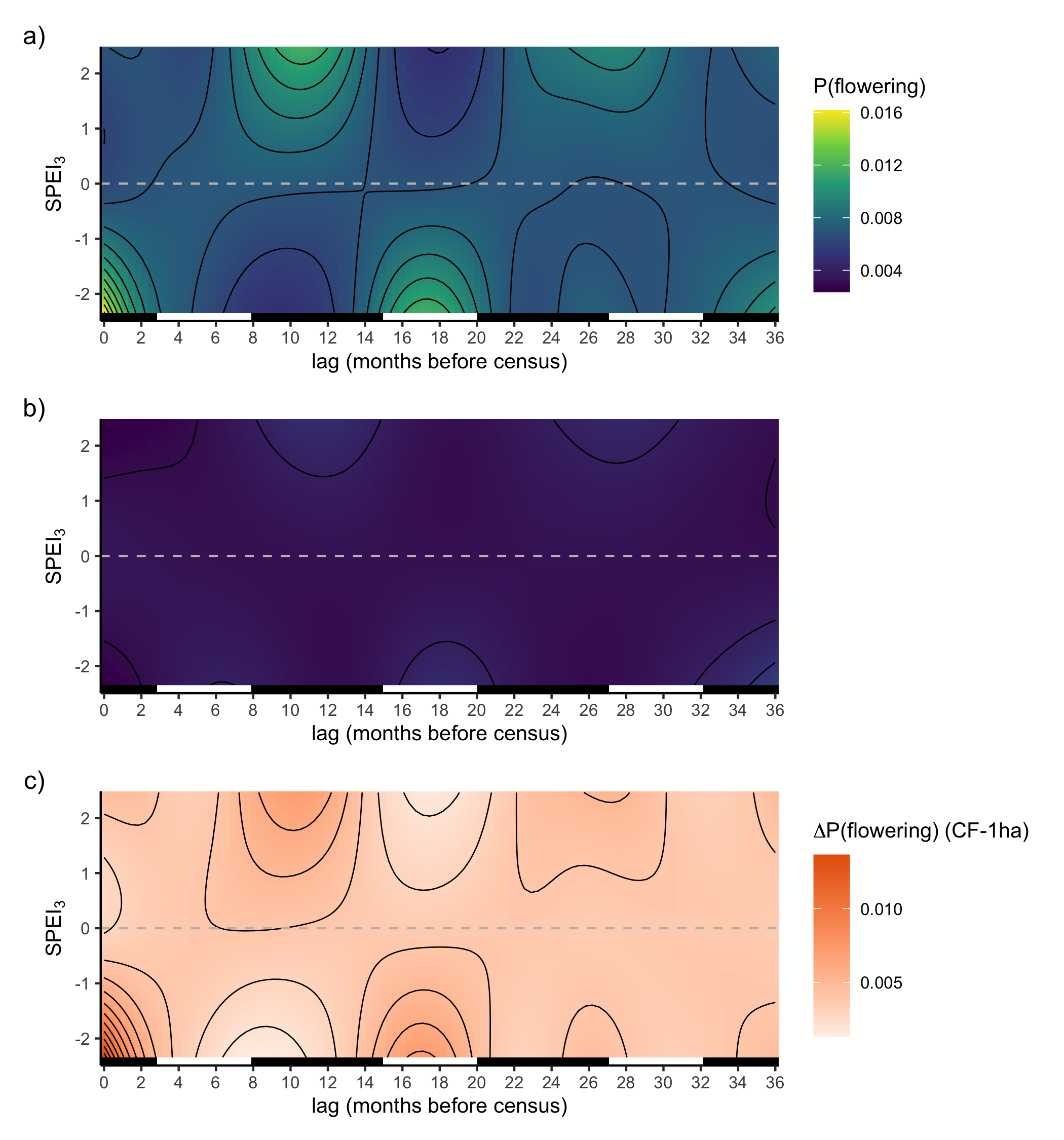


Figure 5: Smooth effect of lagged SPEI on flowering probability for a) continuous forest and b) 1-ha fragments. Panel c) shows the difference between panels a) and b).Surface is modeled as a crossbasis function with cubic regression splines for each marginal basis. Model intercepts were added to fitted values of the crossbasis function and back-transformed to the response scale. Contour lines correspond to a change of 0.001. The bar on the bottom of each panel indicates wet seasons (black, November–May) and dry seasons (white, June–October).

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* populations persisting in fragments may be more demographically resilient to drought than those in primary forest if the individuals susceptible to desiccation died quickly after fragment isolation (Betts et al., 2019).
* While the alteration of microclimate near forest edges is one of the fundamental results to have emerged from the study of fragmented landscapes (Didham & Lawton, 1999; Ewers & Banks-Leite, 2013), little is known regarding how variable it is. It We hypothesize that the microclimate in continuous forest plots is more stable than that in 1 ha fragments, and that this stability will demographically buffers populations in forest interiors (Bruna et al., 2002).
* There can be cascading effects on other species (hispine beetles that depend on Heliconia flowers/fruits, hummingbirds and seed dispersers ( Stiles (1975)
* eg. gene flow and genetic structure among and between plots already shaped by fragmentation ( Côrtes et al. (2013), there can also be drought effects eg Sifres et al. (2007))
* (Janssen et al., 2021) Used machine learning to combine remote sensing and ground surveys of literfall from published datasets. Estimates green-up, literfall, growth, etc. for central and southern America including the Amazon. Drought results in anomalously high leaf flushing at start of drought followed by increased litterfall during the drought. Dry season droughts result in green-up, but not wet season droughts. During drought, old leaves are shed and new leaves are maintained. 2-5 months for newly flushed leaves to fully expand

# Citations / information could allocate to appropriate paragraphs

* Dowd (2020) (unpublished thesis, could write MB and ask for update): precipitation and fragmentation influenced fruit/seed production (conditional on flowering) by *Heliconia* (one step past flowering, so might be worth citing to say that there are other ways reprod. can be affected
* Effects of precip extremes on understory plants can be also be indirect:
  + high precip = lots of clouds, less photosynthesis, less or slower growth
  + low precip = trees lose leaves or die, more light gets through or gaps open. This can be positive (more light = more growth) but not always ( Westerband & Horvitz (2015)) there is a threshold that can lead to photoinhibition ( He et al. (1996), Mulkey et al. (1991)). Drought and herbivory effects can be additive, at least in seedlings ( Barton & Shiels (2020)
* Mulkey & Wright (1996) and Bonal et al. (2016) will have goods stuff; haven’t looked them over yet
* Wright & Calderon (2006): Increase in flower and seed production after El Niño events for trees and lianas. Might be due to increases solar irradiance. Maybe evidence of lagged effects? Table 2 is inconsistent with text, so not sure…
* (Esteban et al., 2021) negative effects of large positive SPEI
* In Manaus, the canopy is as green or greener in the dry season compared to the wet season (Guan et al., 2015), so dry season drought may result in relatively large increases in light penetration to the understory.
* Leitold et al. (2018) increased canopy turnover during drought. Similar increase in continuous and (large) fragments. Increase in turnover more variable in fragments. Coarse woody debris increased 62% during drought.

### Colophon

This report was generated on 2021-05-25 14:13:11 using the following computational environment and dependencies:

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#> system x86\_64, darwin17.0   
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#> P magrittr 2.0.1 2020-11-17 [?]  
#> P MASS 7.3-54 2021-05-03 [?]  
#> P Matrix 1.3-3 2021-05-04 [?]  
#> P memoise 2.0.0 2021-01-26 [?]  
#> P mgcv \* 1.8-35 2021-04-18 [?]  
#> P modelr 0.1.8 2020-05-19 [?]  
#> P munsell 0.5.0 2018-06-12 [?]  
#> P mvnfast 0.2.5.1 2020-10-14 [?]  
#> P mvtnorm 1.1-1 2020-06-09 [?]  
#> P nlme \* 3.1-152 2021-02-04 [?]  
#> P nnet 7.3-16 2021-05-03 [?]  
#> P numDeriv 2016.8-1.1 2019-06-06 [?]  
#> P patchwork \* 1.1.1 2020-12-17 [?]  
#> P pillar 1.6.0 2021-04-13 [?]  
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#> P pkgconfig 2.0.3 2019-09-22 [?]  
#> P pkgload 1.2.1 2021-04-06 [?]  
#> P png 0.1-7 2013-12-03 [?]  
#> P prettyunits 1.1.1 2020-01-24 [?]  
#> P processx 3.5.2 2021-04-30 [?]  
#> P ps 1.6.0 2021-02-28 [?]  
#> P purrr \* 0.3.4 2020-04-17 [?]  
#> P qqplotr \* 0.0.5 2021-04-23 [?]  
#> P R6 2.5.0 2020-10-28 [?]  
#> P RColorBrewer 1.1-2 2014-12-07 [?]  
#> P Rcpp \* 1.0.6 2021-01-15 [?]  
#> P readr \* 1.4.0 2020-10-05 [?]  
#> P readxl \* 1.3.1 2019-03-13 [?]  
#> P remotes 2.3.0 2021-04-01 [?]  
#> P renv 0.13.2 2021-03-30 [?]  
#> P reprex 2.0.0 2021-04-02 [?]  
#> P rlang 0.4.11 2021-04-30 [?]  
#> P rmarkdown \* 2.7 2021-02-19 [?]  
#> P robustbase 0.93-7 2021-01-04 [?]  
#> P rpart 4.1-15 2019-04-12 [?]  
#> P rprojroot 2.0.2 2020-11-15 [?]  
#> P rstudioapi 0.13 2020-11-12 [?]  
#> P rvest 1.0.0 2021-03-09 [?]  
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#> P sessioninfo 1.1.1 2018-11-05 [?]  
#> P snakecase 0.11.0 2019-05-25 [?]  
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#> P statmod \* 1.4.35 2020-10-19 [?]  
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#>   
#> [1] /Users/scottericr/Documents/HeliconiaDemography/renv/library/R-4.0/x86\_64-apple-darwin17.0  
#> [2] /private/var/folders/b\_/2vfnxxls5vs401tmhhb3wqdh0000gp/T/Rtmp2WBGDt/renv-system-library  
#> [3] /private/var/folders/b\_/2vfnxxls5vs401tmhhb3wqdh0000gp/T/Rtmp6ZYiWM/renv-system-library  
#>   
#> P ── Loaded and on-disk path mismatch.

The current Git commit details are:

#> Local: eric-edits /Users/scottericr/Documents/HeliconiaDemography  
#> Remote: eric-edits @ origin (https://github.com/BrunaLab/HeliconiaDemography.git)  
#> Head: [f5a921c] 2021-05-25: add in final citation