Tipping points in neotropical forests: exploring causes, risks, consequences and prevention of large scale...

Technical Report · January 2016

23 authors, including:

Bart Kruijt
Wageningen University & Research

M. Pérez-Soba
Wageningen University & Research

Kirsten Thonicke
Potsdam Institute for Climate Impact Research

Alice Boit
Universität Potsdam

Collaborative Project

Role Of Biodiversity In climate change mitigatioN

Deliverable 2.3.4.
Tipping points in neotropical forests: exploring causes, risks, consequences and prevention of large scale forest dieback

GA number: 283093

FP7-ENV-2011.2.1.4-1

Start date of project: 01 November 2011 | Duration: 48 months

Due date of deliverable: 31/08/2015 [month 46] | Actual Submission date: 28/01/2016

Lead partner for deliverable: Alterra
Authors: Jana Verboom; Bart Kruijt; Marta Perez Soba; Hans Baveco; Michiel Van Eupen; Marielos Pena Claros; Arie Staal; Masha van der Sande; Yao Li (student) - Alterra and Wageningen University, Wageningen UR, The Netherlands
Terry Parr; Laurence Jones; Dario Masante; Beth Purse; Chris Huntingford - NERC Centre for Ecology & Hydrology, UK.
Kirsten Thonicke; Boris Sakschewski; Alice Boit - Potsdam Institute for Climate Impact Research (PIK), Earth system analysis, Potsdam, Germany
Lucieta Martorano, Norma S. Beltrão, Nathalia Nascimento, Eleneide Doff Sotta, Leila Sheila Lisboa, Rodrigo Muniz - Embrapa, Belem, Brazil

Dissemination Level		
PU	Public	Yes
PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the Consortium (including the Commission Services)	
CO	Confidential, only for members of the Consortium (including the Commission Services)	

Publishable Executive Summary

In this report we summarise work on resilience points in tropical forests at regional and site scales in Latin America. In section 1, we review the issue of tipping points and explore the likelihood of climate change-induced or land-use change induced critical change in the neotropical forest ecosystems, based on current literature and recent results from the ROBIN and the related AMAZALERT project. This literature and novel results tend to indicate lower risk of critical transitions than was assumed in earlier work, but the need to monitor potential change is still essential. Although the risk of short-term collapse now appears to be small, there is a large uncertainty which means that the risk still exists. We point out the consequences of such potential degradation in terms of severe loss of ecosystem services.

In section 2, we introduce a mechanistic model that is more realistic (and complex) than most of the simple models used for studying tipping points, yet less complex than the dynamic global vegetation models (DGVM's). The model has a spatial dimension and combines two pressures: climate change (drying) and land use change (deforestation). We use the results of this model to illustrate potential pathways of degradation and providing some ideas about indicators for early warnings of undesirable outcomes. The results indicate synergy between climate change and deforestation: the impact of the combined effect is larger than the sum of the separate effects.

In section 3, we review empirical knowledge on the subject. Analysis of remote sensing data shows that for certain conditions, several states are possible, e.g. dense forest or savannah. These alternative stable states suggest that tipping points do exist. We furthermore show novel unpublished data from Brazil that explore the resilience of the Amazon forest to extreme events such as the El Niño drought of 2005. The results show that although the forest as a whole has recovered from this extreme event, there are changes in time and space that indicate differences in the resilience of the forest between climatic zones.

In section 4 we focus on the effects of habitat loss and fragmentation on connectivity for birds and mammals. We show how future developments in the region could cause loss of patches that are large enough to sustain viable populations of some of the iconic species of the neotropics. We use the jaguar and the sloth as indicator species, representative for habitat specialists that are sensitive to fragmentation.

In section 5, possibilities for monitoring critical change are discussed. We investigate the monitoring options of climate forcing and impact on the main services: carbon storage, water cycling and biodiversity. We also discuss options to monitor socio-economic processes as indicators of change. Novel and future possibilities for monitoring are given special attention. Even if there is much uncertainty over the occurrence of tipping points, it is still worthwhile to undertake this monitoring because, like the canary in the coalmine, it might give us a time window for action before it’s too late and largely irreversible change takes place.

This work builds on the EC’s AMAZALERT project report on “A blueprint for an early warning for critical transitions system in Amazonia”.

Deliverable 2.3.4

Tipping points in neotropical forests: exploring causes, risks, consequences and prevention of large scale forest dieback

Created and maintained by Jana Verboom

Version control	Edited by	Date of revision
Created – V.1.0	Jana Verboom	6/102015
Revised - V.1.1	Laurence Jones	6/10/2015
Revised - V. 2.0	Jana Verboom	12/11/2015
Revised - V.2.1	Terry Parr	19/11/2015
Revised - V. 2.2	Terry Parr	19/1/2016
Revised - V.3.0	Jana Verboom	27/1/2016
Revised Final co-ordinator sign off	Terry Parr	28/1/2016
Revised		

Contents

1. Introduction

Tropical forests are of global importance for various important functions and services to humanity, but global change (climate change and land-use change) is altering these forests possibly even towards a point of no return and a complete shift to another, degraded state.

The aim of this report is to explore the causes, risks, consequences and prevention of large scale forest degradation in neotropical forests. For this purpose, we review existing literature, and introduce a mechanistic model that is more realistic (and complex) than most of the simple models used for studying tipping points, yet less complex than the dynamic global vegetation models (DGVM's). It moreover has a spatial dimension and combines two pressures: climate change (drying) and land use change (deforestation). We use the results of this model for illustration of potential pathways of degradation, pointing out directions of further research, and providing some ideas about indicators for early warning (early warning signals for undesirable outcome, looking for an equivalent of the canary in the coalmine).

We start by defining tipping points and related concepts. In the next sections, we review the current knowledge about such tipping points in neotropical forests. We illustrate the pathways of forest degradation using a simulation model, and provide some novel empirical data to put the results into context. We show some other results that model future changes in two iconic animal species that may result from habitat loss and fragmentation. Last, we end with discussing early warning signals and options for preventing large scale tropical forest dieback.

1.1. Definitions related to tipping points

Complex dynamical systems, ranging from ecosystems to financial markets and the climate, can have tipping points at which a sudden and often unexpected system shift may occur; at a tipping point a small change in the system can have large consequences for the system's state. Such a system change is also referred to as a critical transition, regime shift, collapse, or cascading effects, as when one change causes another change, which triggers another change and so on. When considering the tropical forest ecosystem, the term forest dieback (or specifically Amazon dieback) is used; however, forest dieback can also take place without the presence of a tipping point. Tipping points are a consequence of complex non-linear dynamics, especially positive feedback loops. Feedbacks within a system can either dampen or amplify the response to external factors affecting the system. Negative feedbacks dampen a disturbance by an external factor, whereas positive feedbacks amplify the disturbance effect.

Resilience is the capacity of a social-ecological system to absorb or withstand perturbations and other stressors such that the system remains within the same regime, essentially maintaining its structure and functions. It describes the degree to which the system is capable of self-organization, learning and adaptation (http://www.resalliance.org/resilience). Resilient systems tend to have negative feedback loops that push the system back after disturbance to its old state, or a similar state. Thus, a resilient system can cope with external shocks. When a system approaches a tipping point, resilience declines and the magnitude of a shock from which the system cannot recover gets smaller and smaller.

Because of complex dynamics and feedbacks, many systems can exist in what are called alternative stable states. A stable state is a state that will not change unless it is disturbed by an external factor. The state of a system at any time is defined by the values (amounts) of the variables that constitute the system, e.g., GDP (gross domestic product), % of forest cover and soil moisture content. To help us visualize alternative stable states, the metaphor of a ball in a basin is often used. After a disturbance, the ball will roll back to the lowest point in its basin (resilience). The slope of the basin determines the pull of the lowest point, the stable state. In complex systems, there can be multiple basins. The lowest points are the locally stable states that the system will return to after a small disturbance. The space that leads back to a stable state is called the basin of attraction. However when either the “landscape” is changed so that the slope changes or even disappears, or the disturbance is very large, the ball will leave the basin of attraction and be pulled to another basin (see Figure 1.1). The latter is sometimes referred to as “noise-induced tipping”. When a tipping point of a system is approached, the basin of attraction gets smaller, shallower, and might disappear altogether. Usually, some of these basins are desirable from a human perspective, and others are undesirable, e.g. from a biodiversity and carbon storage perspective rainforest is preferable over savannah. Once a system is in an undesirable state, it is “trapped” there and only a large change (human effort or natural process) – either resulting in a change in the “attraction landscape” or a large disturbance of the state – can get it back to the desirable state.

In the Anthropocene, humans are responsible for a variety of forcing factors, usually referred to as pressures or disturbance regimes (ROBIN indicator framework). Here, we reserve the term disturbance for short-term perturbations of the system, such as floods, landslides (erosion), severe droughts, hurricanes, and fires, and not for long-term pressures such as rising demand for beef, soy and energy crops or climate change. In our metaphor with a ball in a landscape with basins of attraction, the basin landscape and the slopes change due to these external forcing factors of the system. The well-known example from freshwater ecosystems where alternative states are clear water with high biodiversity and muddy water with low biodiversity, has eutrophication as a pressure or forcing factor (Scheffer & Jeppesen 2007).

There are several studies that demonstrate that biodiversity (and/or complexity) enhances stability (for reviews, see e.g., Elmqvist et al 2003, Folke et al. 2004). Response diversity in traits is important because it represents how species respond to disturbances in a species specific way as some are more sensitive to particular disturbances (e.g. drought or flooding) and the more species there are the higher the chance that some will survive a disaster. Functional diversity, or the diversity of species that perform a specific function in the ecosystem, such as pollination, or seed dispersal is also important. The more species that can perform a particular function in the ecosystem, the higher the chance that some will survive a disaster and the function will be maintained. The role of biodiversity has been acknowledged on the basis of experiments, empirical studies and modelling in processes such as pollination, seed dispersal, and maintenance of the seed bank, Elmqvist et al 2003, Folke et al. 2004).

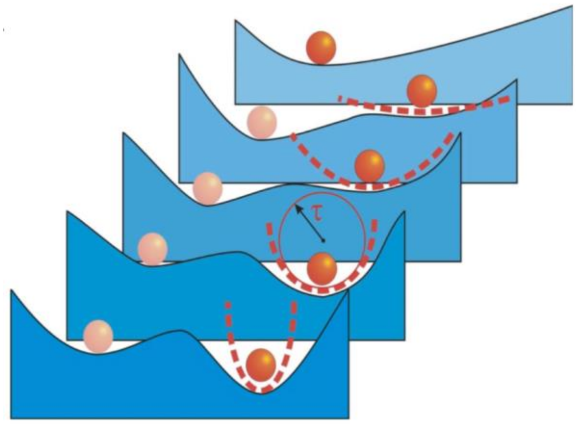


Figure 1.1 Schematic illustration of a tipping point using the ball-in-basin metaphor: the potential basins represent stable attractors, and the ball represents the state of the system. Under gradual anthropogenic forcing (progressing from dark to light blue), the right potential basin becomes shallower and finally vanishes (threshold), causing the ball to abruptly roll to the left. The curvature of the well is inversely proportional to the system's response time τ to small perturbations (reproduced from Lenton et al., 2008). The ball can end up in the other basin in two ways: either the old basin disappears, or the ball is perturbed beyond the basin of attraction due to internal variability: "noise-induced tipping".

1.2. Existence of tipping points in tropical forest systems, forcing factors and causes of change

Neotropical forests are under continued pressure of clearing for mainly agricultural use (Davidson et al., 2012; Aguiar et al., 2013). An estimated 7.3 million hectares of forest are lost each year, and about half of the world's tropical forests have been cleared according to the United Nations' Food and Agriculture Organization (FAO). Several studies have suggested that remaining forests are also under threat from climate change-related pressures, particularly by rainfall reductions (drying) and temperature increase, enhanced by progressive CO2 release to the atmosphere (White et al., 2000, Cox et al., 2000). Moreover, global climate change may affect large-scale patterns of climate variability, which in turn modulate the flow of moisture into Latin America from the adjacent oceans. Models show that the vulnerability of the forests to climate change is aggravated by progressive deforestation (Nobre and Borma, 2009; Staal et al. 2015). These studies also suggest that forest degradation may occur in a highly non-linear fashion, with progressive decline occurring after certain thresholds in global climate or land-use change (i.e., tipping points), have been crossed (Nobre and Borma, 2009). Moreover, whereas most studies concentrated on either climate change or land use change, the combined effect of climate change and land use change may be much more severe than the two pressures alone (Staal et al. 2015, Travis 2003).

From a theoretical point of view it is plausible that alternative stable states do exist in the complex forest-climate system. In this system, at least two different positive feedback loops can be identified: 1) a tree cover-fire feedback which may cause alternative stable states on a local scale, and 2) a tree cover-rainfall feedback which acts on a more regional scale. These two interact and could re-inforce each other: extensive forests stimulate regional water cycling by evaporating moisture (evapotranspiration) and are highly resistant to fire. Logging and deforestation both decrease the local evapotranspiration and increase the risk of fire. As fire causes tree mortality and lack of precipitation hampers tree growth. Therefore, theoretically, two or more alternative stable states can exist, with tipping points (and possibility of critical transitions) between them: Extensive dense forest is a stable state (high humidity, low fire risk) but a state with low-density vegetation (e.g., savannah) is also stable, as forest regrowth is hampered by lack of soil moisture and high fire frequency. If patches are isolated, seed dispersal may be a limiting factor as well (Van Nes et al, 2014), and herbivory and pollination might make the picture even more complex. In temperate forest ecosystems it has been demonstrated that herbivory alone can cause the existence of multiple stable states, as high densities of herbivores such as deer prevent forest regrowth, keeping a site in a grassy state, while in established forests trees are not vulnerable to mammal herbivores (insects are a different story as they can attack trees of all sizes). Only a major change in conditions, or (noise-induced tipping) a major disturbance that affects trees (e.g. a hurricane or fire) or decimates herbivores (e.g. a disease or hunting), can cause the system to shift (Kramer et al., 2003; Schippers et al., 2014). The presence of large carnivores in a system can also affect the system, leading to so-called “landscapes of fear” (Laundré et al., 2001), where herbivory patterns follow the behaviour patterns of herbivores that are adapted to escape predation risk, i.e. spending more time foraging in dense vegetation and less time in open grassy vegetation with important consequences for vegetation patterns. It is unclear whether the effect of wild herbivores and carnivores is large enough to cause multiple stable states in Latin America nowadays. Herbivores in the Amazon however may have another important role, they act as a ‘nutrient pump’, transporting nutrients from (vegetation in) fertile floodplains to less fertile areas. The transportation of phosphorus, which is a limiting nutrient in mature Amazonian forests, has been estimated to be reduced with 98% since the extinctions of megaherbivores in the Pleistocene and may still be ongoing (Doughty et al. 2013). Additional extinctions of large mammals may further decrease the flux of nutrients in the Amazon basin. Reduced nutrient content lowers the ability of forest tree species in sites with low canopy cover, such as disturbed forests, to grow a closed canopy (Hoffmann et al. 2012). Thus, extinction of large mammals may decrease the resilience of Amazonian forests and hamper restoration of degraded lands.

So climate change (mainly drying and extreme events), land-use change (mainly deforestation) and biodiversity loss (mainly loss of pollinators and seed dispersers) can reinforce each other, leading to drying and degradation of the tropical forests (Sampaio et al., 2007, Staal et al. 2015). While fire usually has a more local effect, precipitation and water recycling is a regional phenomenon, and deforestation with reduced evapotranspiration in one place can cause reduced rainfall patterns elsewhere (Avissar and Werth, 2005).

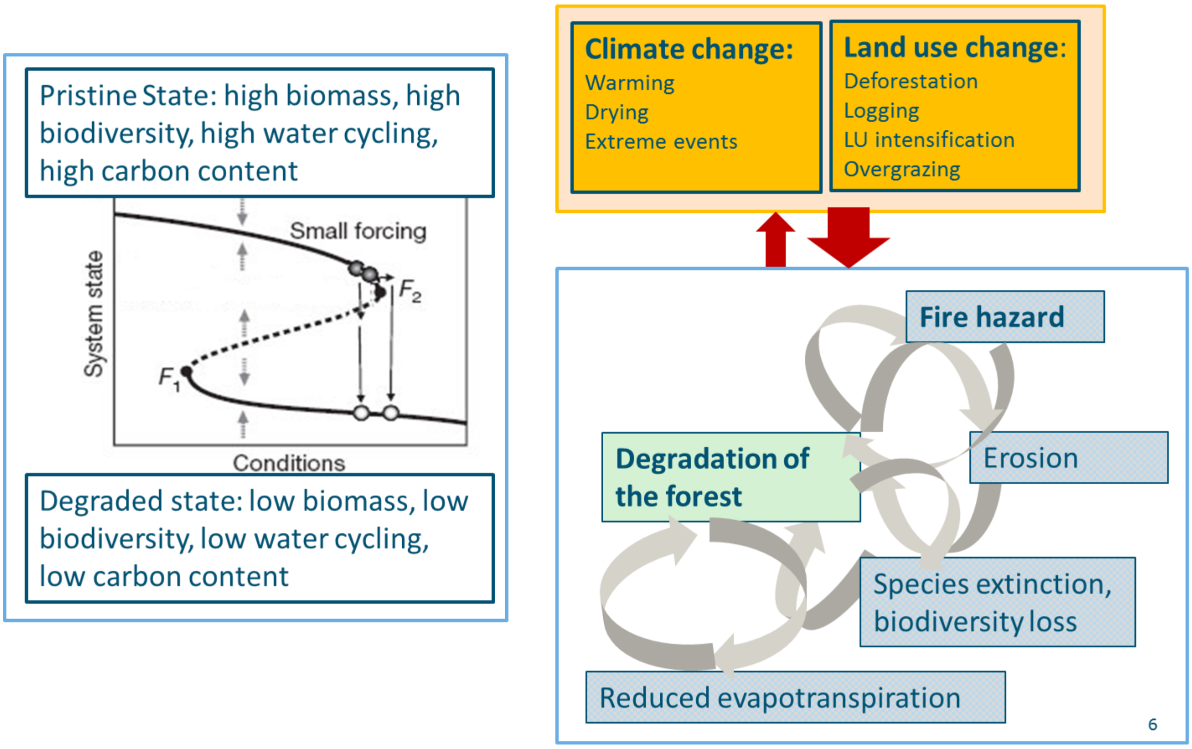


Figure 1.2. The tropical forest ecosystem as a complex system. Figure A shows a theoretical tipping point diagram where a system state can undergo a critical transition when either a threshold condition, or tipping point, is reached (F1, F2) or a disturbance brings the system out of its range of attraction. Figure B illustrates how pressures (climate change, land use change) relate to tropical ecosystem features: there are significant feedback loops such as drying causing forest degradation causing more drying as reduced evapotranspiration leads to reduced rainfall, and degradation leading to fire hazard while fire leads to more forest degradation. Note that arrows go both ways (Figure A modified after Scheffer 2009).

1.3. Risks (how probable is a collapse of a tropical forest ecosystem?)

Because tropical forests are complex systems with many external and internal, highly non-linear processes operating simultaneously on different temporal and spatial scales, there is still a lack of understanding of the exact causes and consequences of climate change, deforestation and forest degradation (Kay et al., 2014; Kruijt et al. 2014). Global climate change, with changed temperature and/or moisture regimes, in synergy with deforestation and logging, possibly amplify feedbacks in the system (associated with water recycling and fire, and possibly also pollination, herbivory and seed dispersal) that are potential forcing factors of forest degradation. Whether these lead to an externally forced forest decline (as in Cox et al., 2000) or a cascading decline caused by feedback loops and transitions across tipping points between alternative stable states (eg, Lenton et al, 2008; Oyama and Nobre, 2003; Lapola et al., 2009) is a matter of debate. It has been shown that in complex models relatively small changes in climate (temperature, dry season length) can indeed eventually lead to large changes in the forest fraction in the Amazon (Good et al., 2011). Recent studies, however, suggest that Amazon dieback is not typical of current global coupled climate-vegetation models and highlight a large lack of understanding of key drivers of change, including the effects of temperature and CO2 concentrations (Huntingford et al., 2013, Good et al., 2013) and the effects of extreme droughts (Meir and Woodward, 2010). Kay et al. (2014) state that “The probability of climate-driven Amazon dieback occurring by the end of the century is significantly less than the probability of it not occurring. However, missing processes and biases (known and potential) in climate and earth system models are such that dieback is much harder to rule out than implied by these models alone. Further, the interactions between climate variability and change and land use change, particularly through fire, are likely to increase the probability of forest degradation”.

Most of what we know about the risk of forest collapse is based on simulation models, although there is also some evidence from remote sensing data. Only models can help us look into the future and answering “what if” questions. However, models are always a simplification of reality. Using them can lead to general insight into the web of causes and consequences, or models can be used to compare scenarios, where we expect that the differences between scenarios are more robust to model assumptions than the absolute outcomes of the model (Verboom & Wamelink 2005). Complexity is not necessarily good, because the more complex the model becomes, the more difficult it becomes to parameterise, calibrate and/or validate it. Complex models tend therefore to have high levels of model uncertainty. We expect that taking more aspects of the system into account could lead to either a more resilient or a less resilient system, because adding relations would introduce both positive and negative feedback loops. The negative loops tend to make a system more resilient, whereas the positive loops would make it less resilient – although of course a feedback loop that increases the stability of one of alternative stable states decreases the relative stability of the other state, so, the same feedback loops that make the preferred state more resilient can make recovery from an unwanted regime shift more difficult. In summary, introducing complexity can either dampen or amplify the response to forcing factors. Additionally, some processes that are not even known to be relevant yet, could become important as the state of the system shifts: we should “expect the unexpected” (Folke et al. 2003).

In summary, although we yet do not fully understand the risk of large scale forest dieback, large old-growth forests seem to have a rather low risk of collapse although the risks are not completely absent. Moreover, as the consequences of large scale dieback would be large and multi-faceted, it still justifies taking it very seriously, both as a scientific community and as policy makers.

1.4. Consequences for ecosystem services and human wellbeing

What would be the consequences of large sections of the neotropical forests turning into grassland with a sparse tree cover, with the number of plant and animal species being severely reduced? Tropical forests deliver humanity a multitude of ecosystem services: provisioning services such as food, wood (for fuel and building) and water; regulating services such as erosion prevention and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services, such as nutrient cycling and pollination. Important and well-known specific services of neotropical forests are, for example, the maintenance of water recycling and transport across the South-American continent (Spracklen et al., 2012), storage and sequestration of substantial amounts of atmospheric carbon (Global Carbon Project, 2014, Gatti et al., 2014) and provision of habitat to a large fraction of global biodiversity (Mittermeier et al., 2003). Many other services and benefits to society can be derived from, or are highly correlated with, these basic, large-scale services; e.g. the loss of biodiversity-dependent ecosystem services is likely to accentuate inequality and marginalization of the most vulnerable sectors of society, by decreasing their access to basic materials for a healthy life and by reducing their freedom of choice and action (Diaz et al., 2006). The existence of most of these ecosystem services ultimately depends on the presence of resilient high-biodiversity, high-biomass, forests in the region. As we have seen in the previous sections, these forests are under pressure from a variety of factors and there is evidence that tipping points between alternative stable states exist. Although a short term (within this century) large scale collapse of e.g. the Amazon rainforest seems improbable at the moment, the consequences for all the services mentioned above would be large even if only a part of the tropical forests was affected. Moreover, there would be even more undesirable consequences for society because of the interconnectedness of systems. The disruption of water recycling and transport across the South-American continent and beyond would lead to changes in the global climate system. Instead of a carbon sink the degraded areas that used to be forest could become a carbon source, further accelerating the climate change process. A significant part of the worlds' biodiversity would be lost. There would be consequences for provisioning services such as forest-originated food, wood (for fuel and building) and water; regulating services such as erosion prevention and disease control; cultural services such as spiritual, recreational (eco-tourism!), and cultural benefits. Most of these consequences would be negative for humanity and could have further social, economic and/or political consequences, potentially leading to social, economic and/or political tipping points. If the system degradation would not be too severe, and some of the basis supporting and regulating services would still function, forests could be replaced by drought resistant perennial crops that have some of the characteristics of forest (e.g., erosion prevention, some level of water cycling). In that case there may be some benefits of the forest degradation for agriculture (crops, livestock) and economy, but this would probably lead to increased inequality and associated socio-economic problems (Diaz et al., 2006).

2. Tipping point modelling results in ROBIN

2.1. Introduction

The possibility of critical transitions occurring in ecosystems is usually explored using simple models, consisting of only a few differential equations. The assumptions of such models are broad simplifications of a complex reality (eg Van Nes et al., 2014, Staal et al., 2015). On the other hand, data-hungry and complex dynamic global vegetation models (DGVM's) exist that can be coupled to climate models, to enable more realistic projections.

The complex dynamic vegetation models do not capture the feedback loop between trees and precipitation, and usually do not take into account deforestation, nor the positive feedback between fire and grasses that may maintain a low canopy cover, and therefore it does not come as a surprise that they do not show tipping point behaviour (Baudema et al. 2015). Most of the coupled climate-vegetation earth system models, however, don't show such tipping point behaviour either. The climate model coupled to a simplified DGVM which has famously projected an Amazon dieback scenario is the HADCM3/TRIFFID model (Cox et al. 2000). This paper triggered many subsequent studies (White et al., 2000; Cox et al. 2004, Cox et al., 2013). Below a certain humidity threshold, regional water cycling is no longer sufficient to support rainforests. Together with temperature increase, this is one of the governing processes in the simulations by Cox et al (2000), where climate change triggered an Amazon dieback. Complicating this issue, vegetation-atmosphere feedbacks influence the local climate because the Amazon forests recycle precipitation through evapotranspiration thus contributing to local rainfall (Zemp et al. 2014). Recent sensitivity studies of vegetation models (Malhi 2009, Lapola 2009, Huntingford et al. 2013; Galbraith et al., 2010) show that the route towards dieback in the models strongly depends on the still very uncertain CO_2 fertilisation effect on tropical tree growth (Rammig 2010). Moreover, drought sensitivity is poorly represented in the models. Empirical data, on the other hand, show that the 2005 and 2010 Amazon droughts have increased tree mortality and reduced tree growth due to water stress (Lewis et al. 2011). While it is still uncertain how the balance of these sensitivities will change when more data become available, it appears that a basin-wide Amazon dieback is still a possible but unlikely future for the Amazon rainforests (Good et al. 2013). However, adding land-use change might lead to other outcomes. In a modelling study including deforestation scenarios, a critical tipping point has been identified at around 40% deforestation when vegetation-atmosphere feedbacks may aggravate water stress (Sampaio et al. 2007). High rates of deforestation would also contribute to increasing fire occurrences, leading to another positive feedback loop (Silvestrini et al. 2011). Recent high resolution data analysis of area burnt and land-cover has shown that fire is now more concentrated in managed pastures than in deforested areas and contributes to forest fires by fires escaping from pasture into neighbouring forests (Cano et al., JGR-Biogeosciences in press) which has implications for future fire projections being more decoupled from deforestation trajectories.

Here we present some tipping point modelling results from simple models to illustrate the possible mechanisms and pathways of forest deterioration. We embed the tri-stable tree-cover dynamics model (van Nes et al., 2014) in a simple, atmospheric water content model, an advection-diffusion model taking into account impact of orographic lifting and seasonality on precipitation, and impact of soil water content and tree-cover on evapotranspiration. The combined model allows us to investigate the dynamics of a system in which local sites with alternative stable states in tree-cover are linked regionally by atmospheric water transport processes. The main questions we deal with are 1) to what extent deforestation may lead to added loss of forest from cascading effects (i.e. a feedback loop where deforestation causes more deforestation), and whether or not such added losses may increase the risk of a regional catastrophic collapse (forest die-back), and 2) how do we expect this will change under a changing climate, with different boundary conditions (atmospheric water content above the ocean) and air transport patterns (wind fields)?

2.2. Exploring the consequences of deforestation in a coupled tree-cover and atmospheric water content model for Latin America.

Baveco, J.M., C. Huntingford, A.G.C.A. Meesters, A. Staal & J. Verboom

Introduction

It is increasingly being recognised that tropical ecosystems can be in alternative stable states. Under a range of precipitation conditions, tropical forest and savannah can be bistable, as is indicated by systematic analyses of remote sensing data of tree cover (Hirota et al., 2011; Staver et al., 2011). Multimodal distributions of relevant variables such as tree cover can be an indication of the presence of alternative stable states (Scheffer & Carpenter, 2003). Therefore, relations between precipitation and the probability distribution of tree cover have been used to assess under which conditions critical transitions of forest loss may occur (Hirota et al., 2011). Because dynamics of systems with alternative stable states can be studied using simple models, the multi-modal tree-cover distributions inspired the development of several simple models aimed at capturing (Van Nes et al., 2014; Staal et al., 2015) and explaining (Staver & Levin, 2012; Good et al., 2015) the inferred alternative stable states in the tropics. Van Nes et al. (2014) constructed a simple model with alternative stable states in tree cover that fits the observed relations between mean annual precipitation and tree cover across the tropics. Depending on precipitation, tree cover can be in a treeless state, a low-tree cover savannah state and a closed-canopy forest state. However, the tri-stable model does not account for an important feedback in large forest ecosystems that may exacerbate multi-stability in tropical ecosystems, which is a regional feedback between tree cover and precipitation. Forests contribute to evapotranspiration by the ability of the trees' deep roots to access deeper soil water than non-forested ecosystems can, implying that deforestation could affect regional precipitation (Nobre et al., 1991; Bonan, 2008). The important role of precipitation implies that the dynamics of forest sites cannot be considered only locally and in isolation, as dynamics are coupled regionally through atmospheric (water) transport processes. However, the behaviour of large-scale spatial forest systems, coupling many locations that may be in alternative stable states, has rarely been studied. Coupled climate-vegetation models have been used extensively, but they often lack the possibility of alternative stable states in tree cover to emerge.

Here we embed the tri-stable tree-cover dynamics model of Van Nes et al. (2014) in a simple, atmospheric water content model. We developed an advection-diffusion model accounting for the impact of orographic lifting and seasonality on precipitation, and the impact of soil water content and tree cover on evapotranspiration. The combined model allows us to investigate the dynamics of a system in which local sites with alternative stable states in tree cover are linked regionally by atmospheric water transport processes. We apply the coupled model to tropical Latin America and explore to what extent deforestation may lead to added loss of forest from cascading effects, and whether or not such added losses may increase the risk of a regional catastrophic collapse (forest die-back).

Material and Methods

Tri-stable tree-cover model

To model tree-cover dynamics we apply the simple 'tri-stable' model of (van Nes et al., 2014). It is defined by the single ordinary differential equation:

$$\frac{dT}{dt} = \frac{P}{h_P+P} r_m\, T \left(1 - \frac{T}{K}\right) - m_A T \frac{h_A}{T+h_A} - m_f\, T \frac{h_f^p}{h_f^p + T^p} \qquad (1)$$

The basis of the model is the logistic growth equation, in which the growth rate r of tree-cover T (%) is a function of the precipitation P. Two non-linear mortality terms are added to this model. One is a weak Allee effect, adding a loss term (mA) at low tree densities (which can also be interpreted as a reduction of the per-capita growth if there are few trees). A similar term is added for fire mortality which decreases steeply above a certain tree density (hf). In the sigmoidal (Hill) function, the exponent p sets the steepness. Growth rate is assumed to be a saturating function of precipitation, with a maximum growth rate of rm. Default parameter values, obtained by calibration/tuning, are listed in table 1 (from Table 1 in (van Nes et al., 2014)).
The behaviour of this model is well-understood. For the default set of coefficient values (Table 1), at each precipitation value either 1 or 2 stable states are possible. With low precipitation only the tree-less state exists, and with high precipitation only the full forested state, while in the intermediate range combinations of tree-less &

savannah and savannah & forest occur. With the default set, there was no precipitation value for which all three states potentially coexisted.

When incorporating feedback between tree-cover and precipitation, (van Nes et al., 2014) noted two main qualitative differences in behaviour. Firstly, the range of climatic conditions (defined by Pd) over which three alternative states coexist was enlarged. Secondly, direct transitions from full forest to treeless conditions (at low rainfall) and the reverse (at high rainfall) became possible.

Atmospheric water content model

We obtained a stationary solution to the water-balance problem for Latin America, under the following assumptions:

The atmospheric water content C (column-value in mm) is prescribed above the oceans, but has to be solved above the continent;

The velocity field is prescribed (constant in time).

Precipitation is parameterized using C, with modifications depending on season, latitude, and terrain slope and height.

Evaporation is parameterized, depending on leaf-area index (LAI).

For ground water a prognostic equation is assumed, containing also runoff.

With these assumptions, it is possible to find the stable fields by a simple relaxation. Necessary input data (maps) include the land-sea-mask and surface elevation. In combination with the tree-cover model, the atmospheric water content model provides local precipitation values (P), while the tree-cover model provides local tree-cover converted into LAI.

Balance equations

For the atmospheric water content C (unit mm), the balance equation is

$$\frac{\partial C}{\partial t} = -u\frac{\partial C}{\partial x} - v\frac{\partial C}{\partial y} + D\left(\frac{\partial^2 C}{\partial x^2} + \frac{\partial^2 C}{\partial y^2}\right) + E - P$$

with D (diffusion, m2/s) a tuning parameter (this is a big number for coarse resolution). For the soil water content W (unit mm) the balance equation is

$$\frac{\partial W}{\partial t} = P - E - W/\tau$$

Herein, τ is a relaxation time for the run-off (presently 300 days; the sensitivity to this will be checked).

Parameterization of evaporation

The evaporation E (unit mm/day) is assumed to depend on a saturated value E0 and on W, according to

$$E = \frac{E_0}{1 + e^{-(W-100)/20}}$$

This yields the optimum value for large W, and zero for very small W. The optimum E0 is assumed to depend on the LAI according to

E0 = Emin + (Emax – Emin) * tanh(cLAI * LAI)

Herein, Emin = 1 mm/day, Emax = 5 mm/day, and cLAI = 0.3.

Parameterization of precipitation

P is assumed to depend on the atmospheric water content C, but also on the distance to the inter-tropical convergence zone (whose location is itself season-dependent), and on orographic lifting. The formula is

P = Pref (1+fh) flat (1 – exp(-bp C))

Pref = 6 mm/day, and bp = 0.05 mm-1. Factor fh is for orographic lifting:

fh = pa a + ph h

Herein, a (unit ms-1) is the ascending wind velocity caused by sloping of the surface; the expression for this is

a = u ∂h/∂x + v ∂h/∂y

in which h is the elevation of the surface in m. The corresponding tuning parameter is pa. If the grid is so fine that it resolves all the slopes, this is enough; but for a coarse grid one needs to account for additional precipitation caused by non-resolved slopes. This problem has been solved in a very easy way by adding the mean local elevation h (in m) multiplied with a tuning parameter ph (unit m-1, this will be a small number). Finally, the dependence on latitude and season is parameterized with factor flat :

flat = 1 – plat (1 – cos(0.12 r))2 /2

in which plat is a tuning parameter, and r is the dimensionless distance to the latitude of the ITCZ (inter-tropical convergence zone), in degrees. The latter latitude is assumed to oscillate between -10 and +10 degrees from the equator.

Coupled system dynamics

Stationary states for the atmospheric water content model were calculated by running the model with a small time step (e.g., 0.005 day) for large number of steps (ca. 5000). During this simulation, wind fields, the boundary conditions for the atmospheric water content, and vegetation cover (obtained from the tree-cover model) were kept constant. The resulting stationary distribution of precipitation P was then used in the tree-cover model, allowing the coupled model to proceed to the next time step. Time step for the coupled model was thus 0.5 year, using January and July data (wind fields, total water column boundary conditions) alternatingly in the calculation of the stationary states of the atmospheric water content model.

Input maps

Historical data, in particular wind fields, were obtained from the ERA INTERIM Reanalysis data set. Downloaded from http://apps.ecmwf.int/ were the data for January and July for the last 10 years in the ERA40 dataset (1993 to 2002), as monthly means of daily means, at 0.5° resolution. Values for east-west and south-north wind components U and V (m s-1) referred to values at pressure level 925 (hPa), roughly at a km above sea level. Static input maps, the land-sea mask and surface elevation, come also from the ERA 40 database.

We compared our results with tree-cover data from the MODIS Vegetation Continuous Fields collection 5 product at 250 m resolution for 2002 (DiMiceli et al., 2011). We averaged these tree-cover data at 0.5° resolution. For comparison with historical precipitation, the same dataset as used by Hirota et al. (2011) was obtained (CRU TS2.1) with the monthly precipitation values (mm) averaged over the period 1961 through 2002 (504 monthly values)).

Analysis

We ran the atmospheric water content model with the fixed (2002) tree-cover distribution, to obtain solutions for precipitation (P), atmospheric water content (C) and soil water content (W) under constant conditions, distinguishing between the January and the July settings.

With the coupled model we investigated the impact of cascading effects occurring after deforestation. Cascading effects imply in this context that deforestation occurring at one location may affect tree-cover at locations further downwind, through decreased evapotranspiration and less moisture being transported. We tested this with a 1000 year simulation starting with the system in its equilibrium state. We simulated a simple deforestation process, implying that every year a random cell (in state=forest) was selected and converted into one with zero tree-cover. Note that one cell is ca. 50 x 50 km2. To test for hysteresis, the process was reverted, with reforestation occurring in the same but reversed order. In the reforested cells, tree-cover was set to its carrying capacity.

Results and Discussion

For constant conditions (current tree-cover), the resulting maps of P, C and W are displayed in figure 2.1.
With deforestation and reforestation, the cascading effects appeared to be minimal (Figure 2.2). Only a few forest cells were lost in addition to the ones removed by deforestation. The initial situation is one with quite a lot of cells in forest state (2317). Removal of less than half of this amount does apparently not lead to indirect losses from moisture cascading. In a previous study based on a simpler advection-diffusion model (ignoring orographic lifting and the seasonal shifting location of the ITCZ), the initial situation was an equilibrium state with far fewer forest cells. Simulating deforestation (a loss of 600 forest cells from an initial number of 718) in this configuration led to larger impact of cascading (Figure 2.**3** and 2.4). The local bi- or tri-stability was what keeps the cells lost through cascading from returning to forest state. The comparison of the results from the two models suggests that it is likely that severe cascading effects will only occur at a late stage of forest degradation, when most of the forest has already disappeared. Further exploration of the parameter

space, and a more formal comparison of model outcome (Fig. 2.5 top row) with observed patterns in e.g., precipitation, is needed to determine at which phase of deforestation cascading effects of reduced moisture recycling are expected to come into play under realistic conditions.

References

Bonan, G.B., 2008. Forests and climate change: forcings, feedbacks, and the climate benefit of forests. Science 320, 1444-1449.

DiMiceli, C.M., Carroll, M.L., Sohlberg, R.A., Huang, C., Hansen, M.C., Townshend, J.R.G., 2011. Annual Global Automated MODIS Vegetation Continuous Fields (MOD44B) at 250 m Spatial Resolution for Data Years Beginning Day 65, 2000–2010, Collection 5 Percent Tree Cover. University of Maryland, College Park, MD, USA.

Good, P., Harper, A., Meesters, A., Robertson, E., Betts, R., 2015. Are strong fire-vegetation feedbacks needed to explain the spatial distribution of tropical tree cover? Global Ecology and Biogeography, in press.

Hirota, M., Holmgren, M., Van Nes, E.H., Scheffer, M., 2011. Global Resilience of Tropical Forest and Savanna to Critical Transitions. Science 334, 232-235.

Nobre, C.A., Sellers, P.J., Shukla, J., 1991. Amazonian deforestation and regional climate change. Journal of Climate 4, 957-988.

Scheffer, M., Carpenter, S.R., 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends in Ecology & Evolution 18, 648-656.

Staal, A., Dekker, S.C., Hirota, M., van Nes, E.H., 2015. Synergistic effects of drought and deforestation on the resilience of the south-eastern Amazon rainforest. Ecological Complexity 22, 65-75.

Staver, A.C., Archibald, S., Levin, S.A., 2011. The global extent and determinants of savanna and forest as alternative biome states. Science 334, 230-232.

Staver, A.C., Levin, S.A., 2012. Integrating theoretical climate and fire effects on savanna and forest systems. The American Naturalist 180, 211-224.

van Nes, E.H., Hirota, M., Holmgren, M., Scheffer, M., 2014. Tipping points in tropical tree cover: linking theory to data. Global Change Biology 20, 1016-1021.

Tables

Table 2.1. Description and default value of parameters of the tree-cover dynamics model.

symbol	description	default value	unit
h_A	Cover below which the Allee effect acts	10	%
h_f	Cover below which the fire increases losses	64	%
h_P	Half saturation for precipitation	0.5	
K	Maximum tree cover	90	%
m_A	Loss rate for Allee effect	0.15	yr^{-1}
m_f	Loss rate for fire effect	0.11	yr^{-1}
p	Factor in Hill function for fire effect	7	
P_d	Precipitation (without plant feedback)	[0.1-10]	mm d^{-1}
P	Mean annual precipitation	[0.5-5]	mm d^{-1}
b	Vegetation-precipitation feedback	0	mm d^{-1}
r_m	Maximum expansion rate of tree cover	0.3	yr^{-1}
r_P	Maximum rate towards equilibrium for precipitation	1	yr^{-1}

Table 2.2. Description and default value of parameters of the atmospheric water content model. The last four parameters are used as tuning parameters.

symbol	description	default value	unit
E_{min}	Minimum value of evapotranspiration	1	mm d^{-1}
E_{max}	Maximum value of evapotranspiration	5	mm d^{-1}
c_{LAI}	Coefficient for saturation relationship *E* with *LAI*	0.3	m^2 leaves m^{-2} soil
b_P	Sensitivity of *P* on *C*	0.05	mm^{-1}
P_{ref}	Reference value precipitation	6	mm d^{-1}
D	Diffusion coefficient	10^6	m^2 s^{-1}
p_{lat}	Dependency on latitude and season	0.5	-
p_h	Constant for impact of unresolved slopes	$2x10^{-3}$	m^{-1}
p_a	Dependency of orographic lifting on slope	100	s m^{-1}

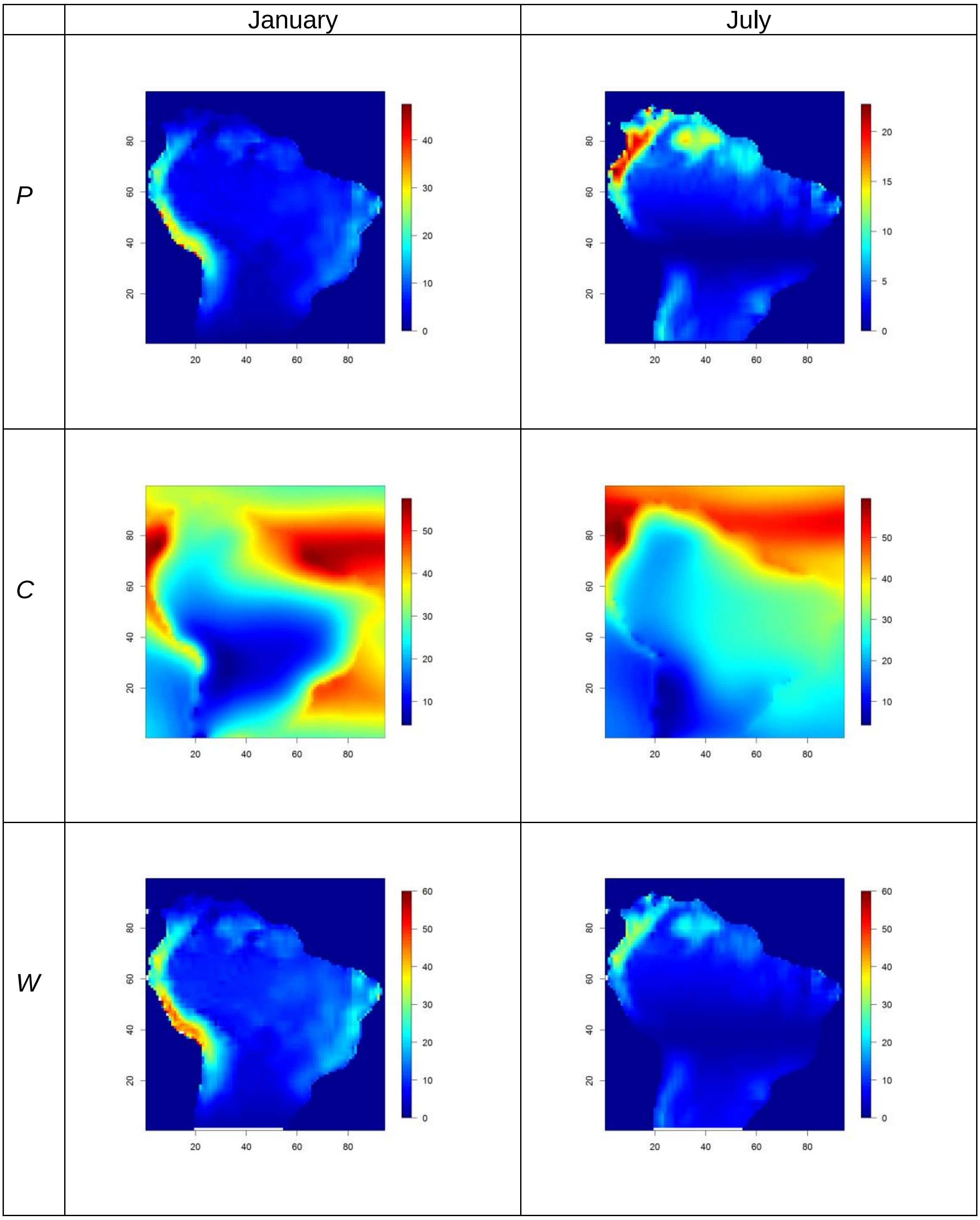


Figure 2.1. Precipitation (P), atmospheric water content (C) and soil water content (W) resulting from solving the advection-diffusion equation for constant tree-cover (2002 MODIS data).

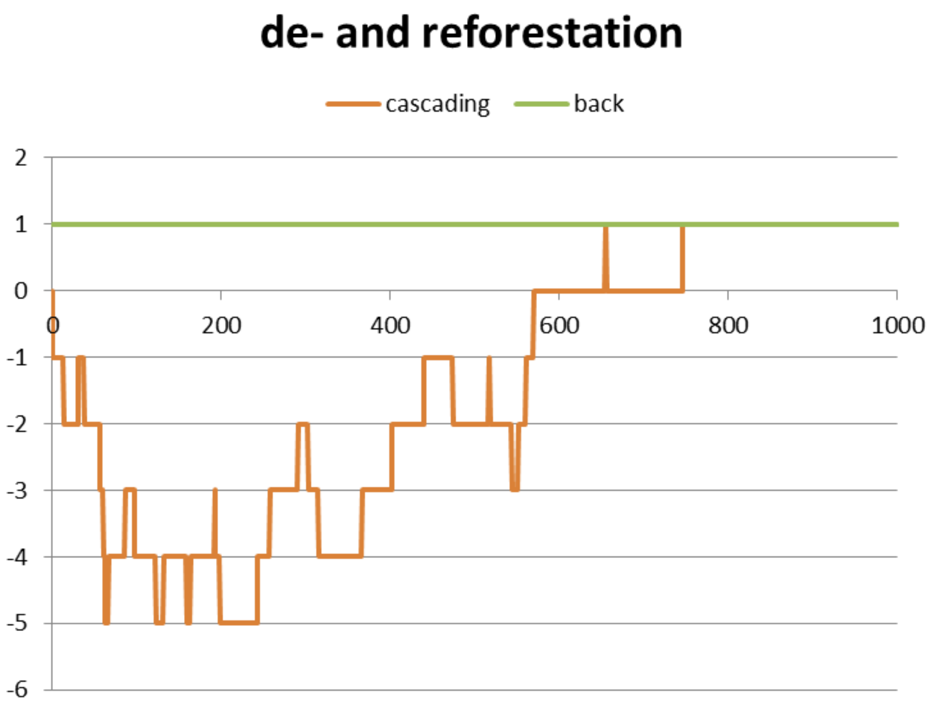


Figure 2.2. The forest cells lost by cascading effects (orange line) are the ones that do not return to forest state when the deforested cells are converted back into forest in exactly the same, reversed order (green line). Simulated period 1000 years (x-axis), with 1 forest cell removed per year.

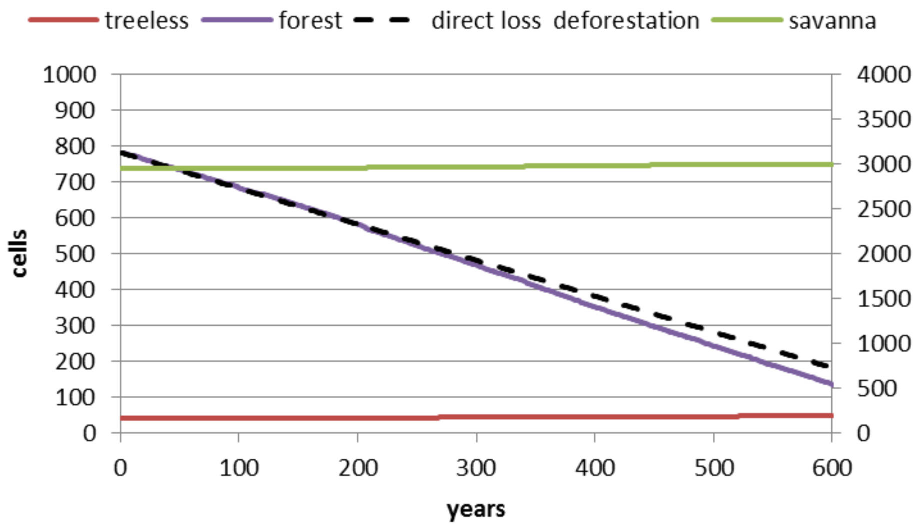


Figure 2.3. With deforestation at a rate of 1 forest cell being converted in a tree-less state per year, the number of forest cells (purple) declines faster than 1 cell per year (dashed line). The additional decrease is due to cascading effects. Results from a simpler model that had at equilibrium a far lower number of forest cells.

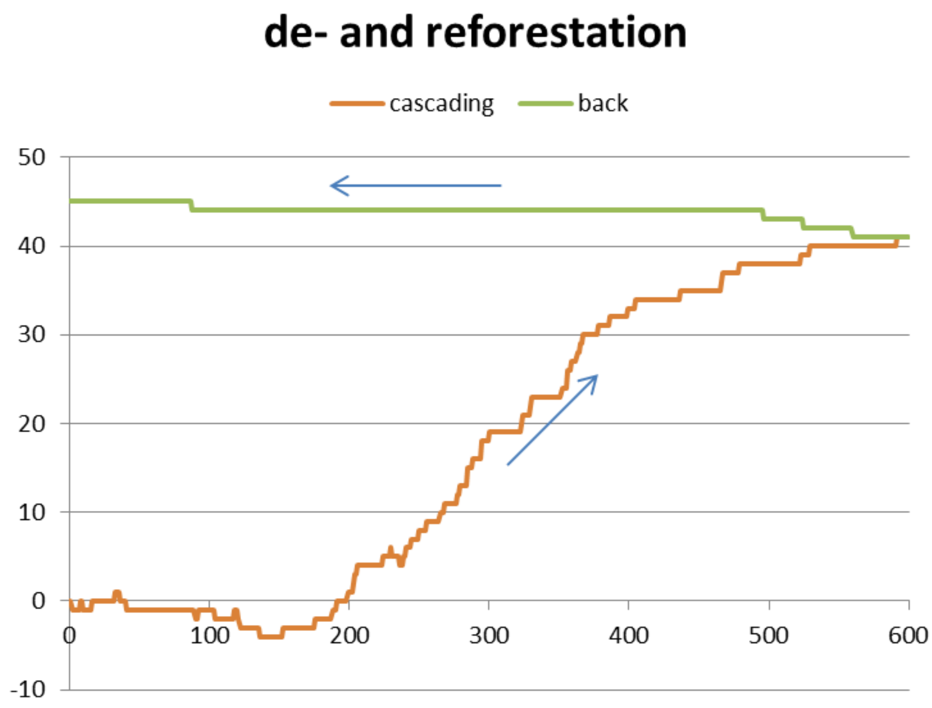


Figure 2.4. As figure 2.2. Results from a simpler model that had at equilibrium a far lower number of forest cells.

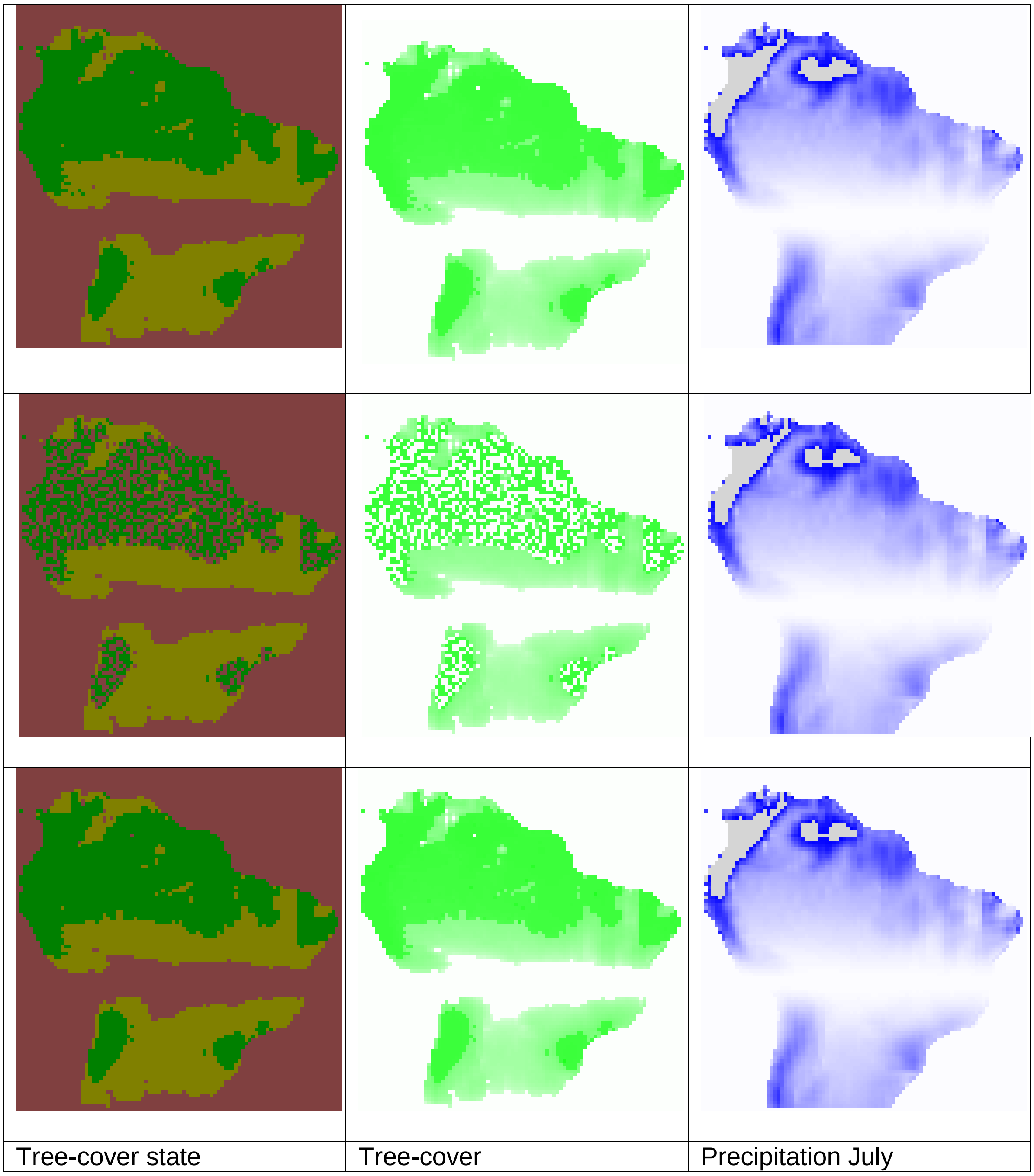


Figure 2.5. Tree-cover state (left), tree-cover (middle) and daily precipitation in July (right) at year 0 (no deforestation), top row, at year 1000 (middle row) and at year 2000 (when deforestation has been completely undone), bottom row.

2.3. *Forest degradation and possible impact on crops*

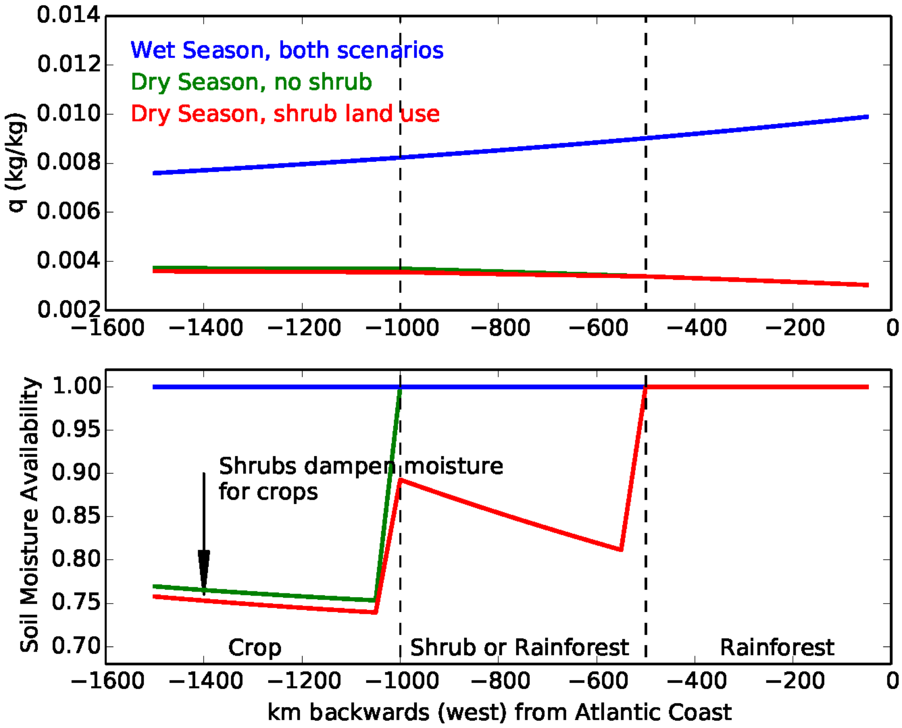


Figure 2.6. We present results illustrating the idea of deforestation affecting crop viability downstream. Consider a column of air above the land surface in a one dimensional landscape, moving with a certain velocity from east (Atlantic coast, right side of the graph) to west (left side of the graph). The x-axis (km) is distance from Atlantic Coast, and the column of air has a humidity q (kg water /kg air). This air humidity is affected by precipitation from the column and evaporation from the land surface. We assume there are two seasons: a dry season and a wet season (blue lines). We consider two scenarios. In one scenario (dry season: green lines) the rainforest stretches from the coast up to 1000 km inland and the adjacent 500 km consist of crops. A second scenario (dry season: red lines) has 500 km of an intermediate degraded vegetation through earlier land use, called shrub, replacing 500 km of rainforest. Evaporation, or evapotranspiration is assumed to be highest in rainforest, intermediate in shrub land, and low in crop land. Figure A shows air moisture, Figure B shows soil moisture. Soil moisture is an important limiting factor for plant growth and survival. The figure shows almost no difference for wet season

and for air moisture, but in the dry season there is a decreased soil moisture in crop land due to replacement of rainforest by shrubs. The mechanism behind this is that shrubs have a smaller rooting depth and recycle less moisture to the atmosphere. This implies less rain even further West, which reduces soil moisture availability for crops. [Red line less than Green line for “Crop”]. NB the dominant wind pattern is assumed East to West. [Chris Huntingford et al. in prep, see appendix 1 for details].

3. Empirical results

Pan-tropical analysis of vegetation patterns derived with remote sensing shows that forest and savanna may both exist under a narrow range of climate conditions as alternative stable states (Hirota et al., 2011, Staver et al., 2011). When two alternative stable states exist, tipping points exist too and this indicates an intrinsic risk for critical transitions in response to relatively small variations in regional climate. Below we present some novel results from the ROBIN project.

3.1. *Evidence of ecological resilience clusters using climate typology in the Amazon rainforest: a methodological approach*

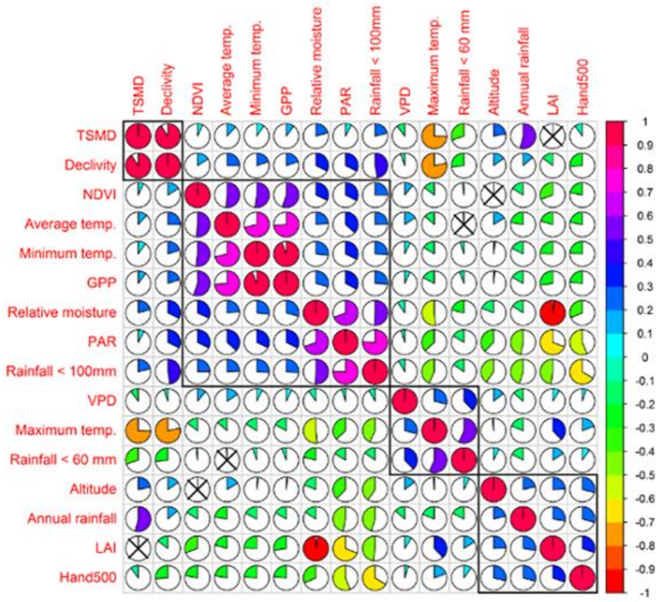
Introduction

Resilience is usually defined as the capacity of an ecosystem to absorb disturbance without shifting to an alternative state and losing function and services (Ives & Carpenter, 2007). The concept therefore encompasses three distinct processes: resistance, the magnitude of disturbance, and the pace of return to the original structure. These processes are fundamentally different but rarely distinguished (Holling, 1996). Increasing the resilience of natural systems may therefore have important implications for human welfare in the face of global climate change. Areas with forest contribute to maintaining humidity, due to evapotranspiration exchanges. In Amazon rainforest, severe droughts and periods with diverse anthropogenic pressures are threatening ecological resilience in the region. This research aims to identify areas with a differentiated resilience capacity in the Brazilian Amazon and to assist management and monitoring strategies in order to ensure preservation in least resilient areas and sustainable use in areas with greater ability to adapt to current conditions and climate change scenarios.

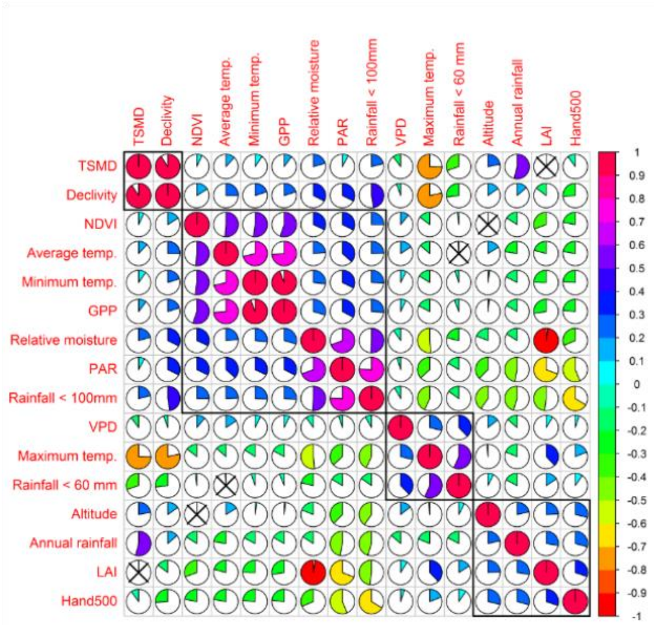
Material and methods

Our approach to measuring resilience was to evaluate the impact of extreme climate events on forest attributes measured by remote sensing. 2005 was an extremely dry year for the Amazon (El Niño). This almost coincided with the year of peak land use as in 2004 deforestation was at its maximum before the Brazilian government passed legislation to protect the Amazon forest. 2009 was a year of strong La Niña with high rainwater supply in the Amazon, and 2013 also had a high rainwater supply. Therefore, 2013 was used as the second time point to analyse forest recovery. Resilience is defined as the ability of Amazon forest areas to recover ecosystem components (such as biomass) lost during periods of extreme shortage of water, as in the case of ENSO meso-scale events. We used biophysical data from the MODIS satellite to assess the temporal response of vegetation in relation to seasonal conditions and variations in weather, climate and altitude (from TOPDATA). The results were then related to maps of climate typology developed using the methodology described in Martorano et al. (1993).

Data integration was performed in ArcGIS by taking into account the correlation analysis among spatial-temporal variables. The variables used in the analysis are shown in Figure 3.1. Since the original data were highly correlated, a Principal Component Analysis (PCA) was used to reduce collinearity and to select which variables should be incorporated in the assessment of ecological resilience. The first component used to generate the maps explained 40.5% and 41.7% of the variance in the data set, in 2005 (Figure) and 2013 (Figure), respectively. By comparing the climate typology ranges with the ranges of response of high correlation variables we established 8 “climate resilience” classes (Martorano et al., in prep.).



2005 (a)



2013 (b)

Figure 3.1. Correlation between variables for 2005 (a) and 2013 (b)

Results and discussion

Table 3.1 shows a cross-tabulation of the 10 climate subtypes in Figure 3.2 (Af1, Af2, Af3, Am1, Am2, Am3, Am4, Aw3, Aw4 and Aw5) with the 8 classes representing differentiated resilience capacities, according to our analysis of the years 2005 and 2013 (Figure 3.3 and Figure 3.4). This shows some clear differences and changes after the severe drought in 2005 (El Niño) indicating some forest zones with a greater ability to recover and zones that are more sensitive to climate variations.

These results are now being prepared for publication and review in a refereed journal.

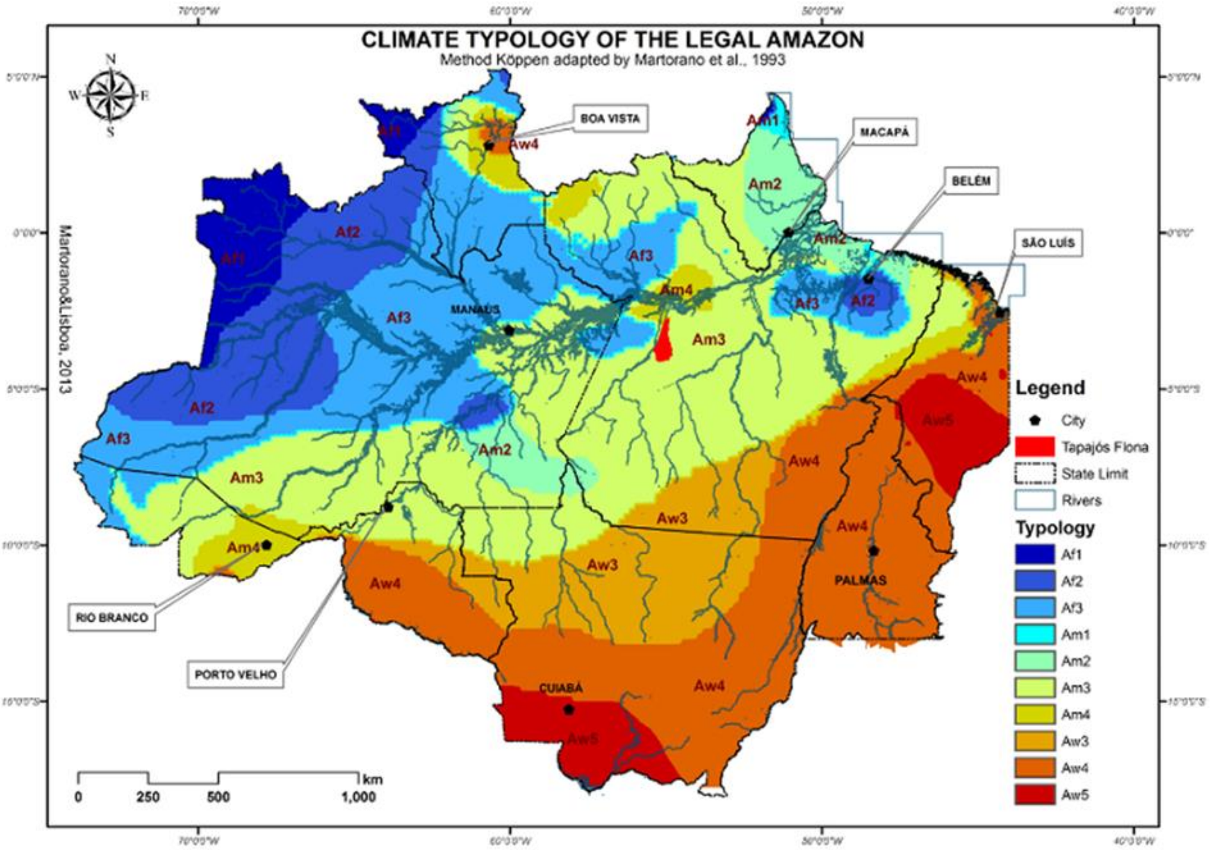


Figure 3.2. Climate typologies in the Amazon.

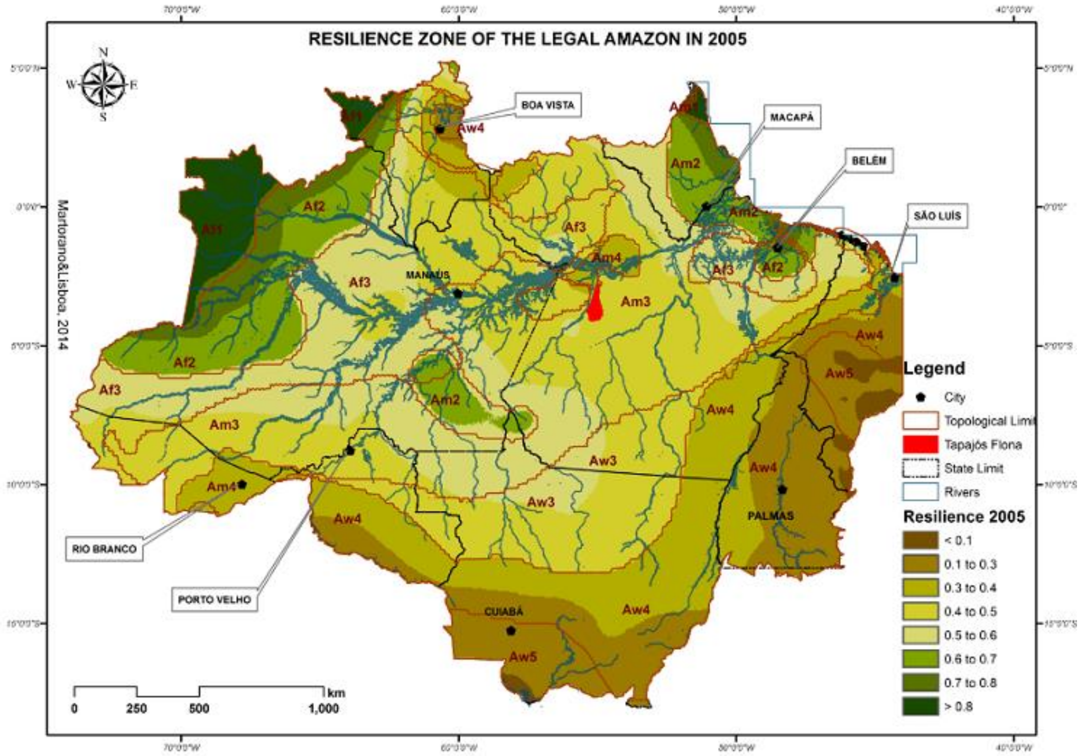


Figure 3.3. Zones with specific ecological resilience capacity in 2005 and the Climate Typologies in the Amazon.

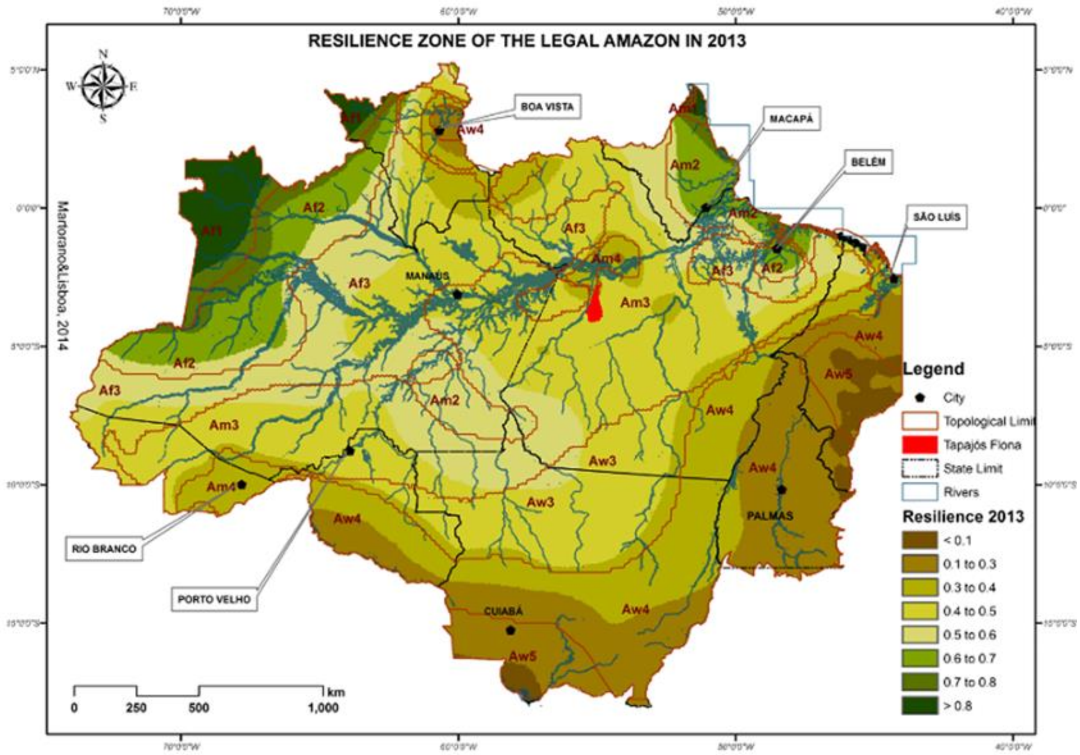


Figure 3.4. Zones with specific ecological resilience capacity in 2013 and the Climate Typologies in the Amazon.

Table 1.1. Resilience zones associated with climate typology zones (see Figure 3.1) in the Amazon. Climate Zones Af1 to Af3 are the most valuable (in terms of carbon stocks and biodiversity) rainforest zones and also the most resilient. The table shows the area (as a % of total area in 2005 and 2013) and the difference between the two. For example, the wettest forest class Af1 has lost 17% in the highest resilience class (>0.8) to the second resilience class (0.7-0.8) between 2005 and 2013.

RESILIENCE CLASS

	< 0.1			0.1 to 0.3			0.3 to 0.4			0.4 to 0.5			0.5 to 0.6			0.6 to 0.7			0.7 to 0.8			> 0.8		
TIPOLOGY	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF	2005	2013	DIF
Aw5	16,7	30,0	13,3	83,3	70,0	-13,3																		
Aw4	1,1	1,7	0,6	48,3	57,5	9,2	50,1	40,8	-9,3	0,5	0,0	-0,5												
Aw3							1,2	20,3	19,1	84,2	75,4	-8,8	14,7	4,3	-10,4									
Am4				2,4	7,2	4,8	92,8	92,8	0,0	4,8	0,0	-4,8												
Am3							1,2	7,6	6,4	59,0	65,1	6,1	38,6	27,0	-11,6	1,2	0,1	-1,1	0,1	0,1	0,0			
Am2										0,0	0,1	0,1	17,2	52,2	35,0	66,7	40,5	-26,2	16,1	7,2	-8,9			
Am1																			17,2	37,8	20,6	82,8	62,2	-21
Af3							2,7	5,4	2,7	37,6	48,4	10,8	58,7	45,3	-13,4	0,4	0,4	0,0	0,3	0,2	-0,1	0,2	0,2	0
Af2													6,6	28,6	22,0	64,5	55,2	-9,3	28,8	16,2	-12,6			
Af1																			9,2	26,4	17,2	90,8	73,6	-17

Conclusions

Based on our definition of resilience, the methodology proposed shows that the response of biophysical variables from satellite information combined with typological climate conditions allowing us to differentiate variations in the Amazon's ecological resilience capacity.

References

Ives, A. R, Carpenter S. R 2007. Stability and diversity of ecosystems. Science 317: 58–62.

Holling, C. S. 1996. Engineering resilience versus ecological resilience. In P. C. Schulze, editor. Engineering within ecological constraints. National Academy Press, Washington, D.C., USA.

Martorano, L. G., Nechet, D., Pereira, L. C. 1993. Tipologia climática do Estado do Pará: adaptação do método de Köppen. Boletim de Geografia Teorética, v. 23, p. 45-46.

4. Effects of landscape change on wildlife

For many wildlife species, it is important to have large patches of habitat, for preventing local population extinction, and a certain connectivity across the landscape, enabling movement between patches and promoting gene flow. Habitat connectivity is thus an important factor for biodiversity conservation, and connectivity can be used as a proxy for biodiversity. Connectivity can be measured and modelled on the basis of maps and scenarios more easily than biodiversity. In this chapter we present two different habitat connectivity results produced by two different models: GLOBIO and GRIDWALK/LARCH.

4.1. GLOBIO results on minimum area requirements for wildlife

Minimum area requirements (MAR) for viable populations can be used in a biodiversity assessment to understand the impacts of fragmentation (habitat loss and reduced patch sizes) on the potential viability of bird and mammal populations. We used the GLOBIO3 model (Alkemade et al. 2009), focusing on bird and mammal species. We applied the dose-response relation between area of unfragmented natural habitat and the percentage of species meeting the standards for a MAR in the area at a global level for both the current situation and a future projection. The model combines current land use from the GLC2000 global land cover data set with a resolution of a 30 arc seconds (approximately 1 by 1 km), and projected land use changes from the IMAGE 2.4 model with a resolution 0.5 by 0.5 degrees (about 50 by 50 km). Together with an overlay of the main roads derived from the Digital Chart of the World (DMA 1992) it resulted in a map of patches of natural areas. Although other GLOBIO3 applications take several habitat quality variables into account (Alkemade et al. 2009), for this analysis we ignored habitat quality, focusing on habitat size and connectivity only.

Future patch sizes (2050) were adapted to reflect the future expected land-use change, by adding or subtracting the amount of natural area assigned to each grid cell. The land use projections were derived from the OECD baseline scenario, which is a trend scenario that assumes minor changes in current policies but includes policy actions agreed upon in different international conventions (OECD 2012). Figure 4.1 shows the results for the neotropics, showing biodiversity loss due to habitat loss and fragmentation. The results show that the estimated fraction of mammal and bird species that may not have sufficient undisturbed land area to maintain viable populations in (A) 2000 and in (B) 2050 under land use change according to OECD baseline scenario, assuming that current tertiary roads would be upgraded to secondary or primary roads and become a barrier to wildlife. The maps reveal regions, characterized by high urbanization and agricultural activities, where a large fraction of mammal and bird species cannot currently maintain viable populations, although these species might still occur in these areas. Between 2000 and 2050 more parts of the region are expected to diminish in their capacity to support the area requirements of species, whereas locally, conditions may improve because of land abandonment. The large undisturbed habitats required by many charismatic bird and mammal species are likely to become scarcer in human-dominated landscapes. The results are mainly linked to the assumption that what are tertiary roads in 2000, and are not assumed to be a barrier, in 2050 will be more busy roads which do cause isolation between habitat patches on both sides (e.g. Bolivia). For a full methodology and discussion, see Verboom et al., 2014.

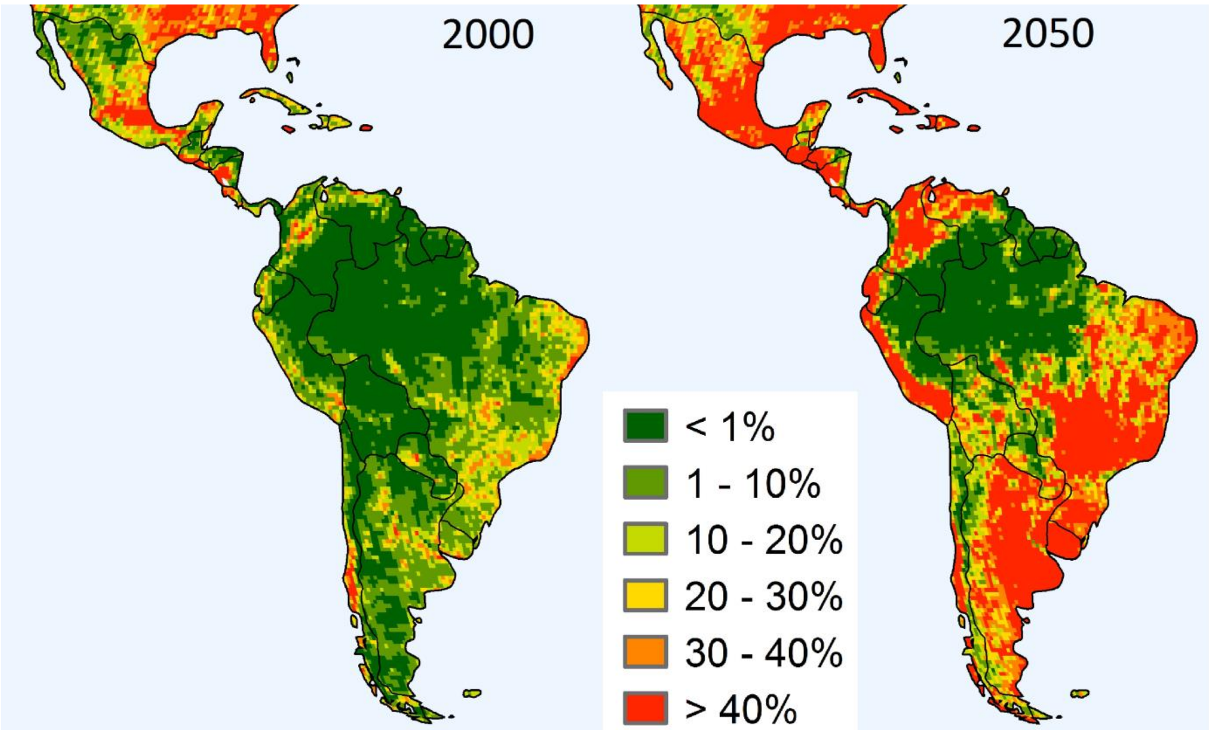


Figure 4.1. GLOBIO connectivity analysis based on minimum area requirements of bird and mammal species, applied to a land use/land cover map from 2000 and the expected map for 2050. The figure shows the average loss of biodiversity due to habitat loss and fragmentation by land conversion and roads. See Verboom et al. 2014 for details.

4.2. GRIDWALK and LARCH model results on connectivity

Whereas the GLOBIO model focusses on patch size for viable populations and ignores the landscape between the patches, GRIDWALK focusses on the connectivity of the entire landscape, including the “matrix”, the part of the landscape unsuitable for long term survival, but necessary for inter-patch dispersal movements. It uses not only a map of suitable habitat, but also a map of permeability to dispersing individuals. It is based on the movements of individuals in a grid landscape, hence the name (gridwalk).

We used two iconic species for the analysis: the Jaguar (Panthera onca) and the Sloth (Bradypus variegatus). The Jaguar is representative for species that need large undisturbed areas for a viable population but can disperse through unsuitable habitat relatively well (large MAR but good disperser). The home range size of the jaguar is estimated to be ca. 30 km2 (Soisalo and Cavalcanti, 2006), and it can disperse to a distance of 50 km. The sloth on the other hand is representative for species that don't need much area for survival, but have very restricted dispersal (small MAR but bad disperser).

We use the Shared Socio-economic Pathways (SSPs) developed within the ROBIN project to describe different socio-economic scenarios (Jones and Kok, 2013). Five socio-economic scenarios are set as: 1) Sustainability (SSP1); 2) Middle of the road (SSP2); 3) Fragmentation (SSP3); 4) Inequality (SSP4); and 5) Conventional development (SSP5). In the end, two Shared Socio-economic Pathways are chosen for use in the ROBIN project, SSP1 and SSP5.

A set of policies are defined, mainly focusing on the land management for mitigating climate change and providing additional safeguards. The policies are: reducing deforestation, reducing degradation, increasing re-afforestation, safeguarding or improving biodiversity and a focus on improving ecosystem services provision. These policies affect the land use. Two policy options are selected, which are: C0 and C3+BD+ES, for simulating the scenarios. Policy option C0 means absence of Carbon-focused or other policies. Under this policy option, no policy is established to manage carbon stocks or additional safeguards, which may be a result of the abandonment of existing policies, or the failure to enforce the existing policies. In contrast, policy option C3+BD+ES refers to the situation, where Carbon, biodiversity and other Ecosystem Services are considered and highly protected.

By combining the Shared Socio-economic Pathways with the policy options, four scenarios are chosen in this chapter, which are: current, SSP1P, SSP5P and SSP5S. SSP1P, SSP5P and SSP5S are future scenarios for 2050, while the current scenario describes the scenario in year 2005. SSP1P means the area is developing in a sustainable way, and full environment protection policy (C3+BD+ES) is implemented. SSP5P and SSP5S describe scenarios when the region is developing in a conventional way. However, under SSP5P, the environment is protected by policies, while no environment protection policy is implemented under SSP5S.

The ROBIN project uses the CLUE (Conversion of Land Use and its Effects) model (van Eupen et al., 2014) to simulate the land use under different scenarios. Sixteen classes of land use types are chosen in the end, for the use of the ROBIN project, among which eight are dynamically modelled by the CLUE model, while the others are expected to be static over time and space. The dynamic land use type classes are: forest; shrub land; grazed shrub land; grassland; grazed grassland; cropland food (feed + fodder); cropland food (perennial tree/shrub); and cropland energy crops. The land use changes are determined by the changes in the demand for different land use types. The demand for each land use type is estimated by country, based on the land use map in 2005. Food balance sheets FAOSTAT was used as a database providing crop production by country by category, and the modellers fitted a linear trend through the production per crop per country to get the production until 2050. In terms of the yields of crops, they were calculated from the FAOSTAT for the period 1961-2011, and were fitted into a linear trend to get the yields until 2050.

A simplified version of the GRIDWALK model is used in this paper to determine the functional connectivity for jaguar and sloth habitats. GRIDWALK model is a widely used grid-based model to assess landscape connectivity (Schippers et al., 1996). The landscape consists of patches (as in GLOBIO) and networks of patches connected by dispersal. We assume that if one or more individuals are expected to move between patch a and b per year, then patches a and b belong to the same network.

The LARCH model (Verboom et al., 2001, Opdam et al., 2006) is then used to decide which patch populations are viable and what patches are linked into ecological networks. LARCH identifies patches with viable populations, i.e. potential population size should be larger than Minimum Viable Population (MVP). A MVP is the minimum population size a species should meet to survive 100 years with a probability of 95% or more. So we calculate the total number of potential territories in a scenario, and the number which is present in large, viable patches. The ratio, or the fraction of area in viable patches, is a measure of fragmentation. If it is one, all habitat is in large viable patches. Below one, habitat is lost due to patches being too small for long term viability. At zero, no single patch would be large enough for a long term viable population. This fraction thus can be used as a proxy for landscape cohesion.

We divided the area in a Central American part (ca), North-East South America (ne), Subtropical (st) and Amazon (am). The subdivision is for technical reasons: the model cannot track millions of animals at the same time. See Figure 4.2.

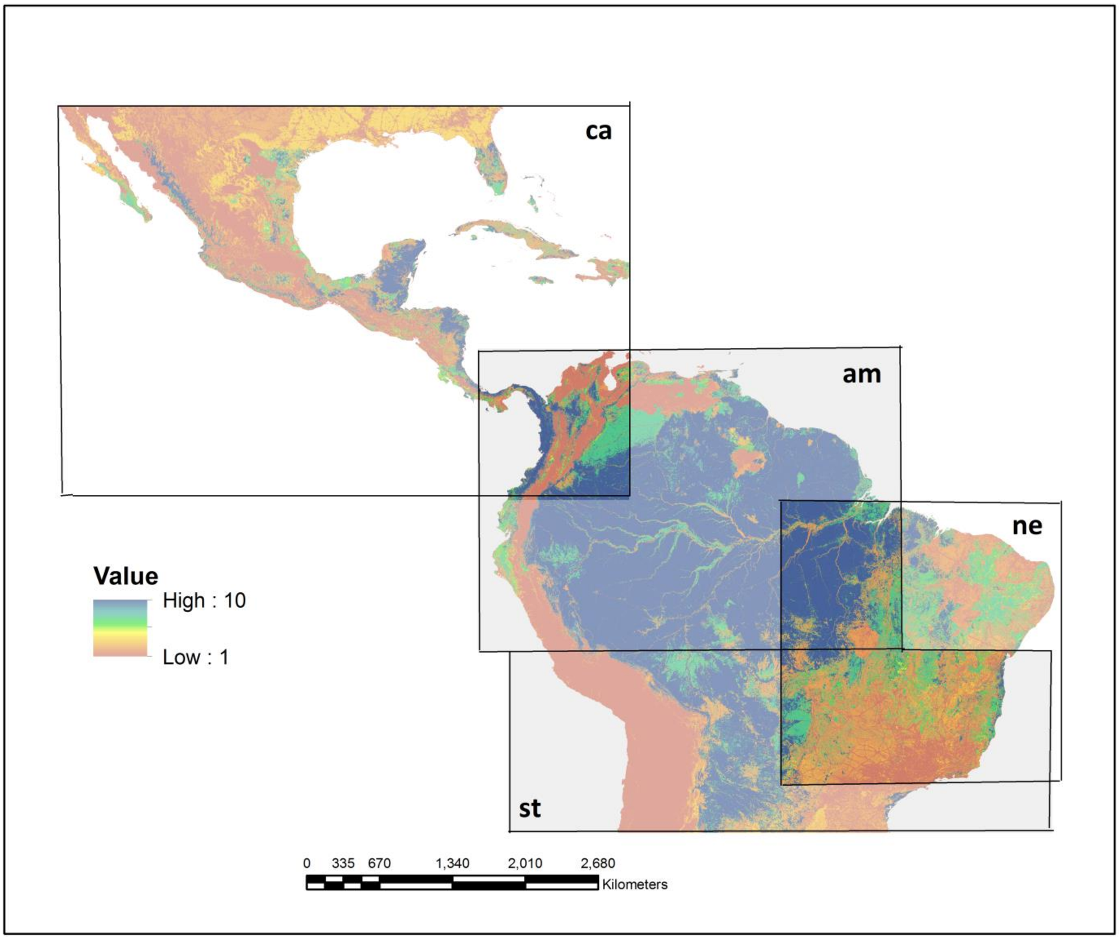


Figure 4.2. Division of the research area into four regions: Central America (ca), North-East South America (ne), Subtropical (st) and Amazon (am). Note that there is overlap. The colour shows the assumed preference value for Jaguar: red has the lowest preference, blue has highest preference.

Tables 4.1 and 4.2 and Figure 4.3 show the outputs of the GRIDWALK/LARCH model. The “number of total reproductive units” indicates the total potential population (pairs) in the landscape, reflecting the total habitat amount. The “number of reproductive units in viable patches" is a more important indicator, since its value reflects the population that can survive for generations. The "proxy of cohesion" is calculated by dividing the "number of reproductive units in viable patches" by the "number of total reproductive units" as a proxy for the cohesion of the habitat. The "number of network" indicates the functional connectivity in the area under each scenario, and is zero if patches are all isolated; and the "number of patches in the network" shows how many habitat patches are functionally connected by dispersal, with an estimated one individual per year exchange, or more.

Table 4.1. Outputs from the GRIDWALK model for the jaguar under different SSPs (Shared Socio-economic Paths) and policy options.

Scenario and region	Number of reproductive units in viable patches	Number of total reproductive units	Proxy of cohesion	Number of networks	Number of patches in the network
current_ca	2831.5461	3407.7472	0.8309	0	-
SSP1P_ca	2667.9389	3368.5897	0.7920	0	-
SSP5P_ca	2378.7286	3034.0507	0.7840	0	-
SSP5S_ca	2374.8312	3020.4806	0.7862	0	-
current_ne	6303.9117	7064.6878	0.8923	1	2
SSP1P_ne	6101.5326	6858.2878	0.8897	1	5
SSP5P_ne	4672.2270	5569.5232	0.8389	0	-
SSP5S_ne	4580.9622	5405.4798	0.8475	0	-
current_st	5199.2175	5684.8476	0.9146	0	-
SSP1P_st	4827.1226	5344.9365	0.9031	0	-
SSP5P_st	3685.3621	4346.3753	0.8479	0	-
SSP5S_st	1595.0535	4247.0557	0.3756	0	-
current_am	21766.4732	22235.1748	0.9789	1	9
SSP1P_am	21359.9959	21868.1377	0.9768	1	10
SSP5P_am	19120.4087	19709.7379	0.9701	1	5
SSP5S_am	19120.4087	19709.7379	0.9701	1	5

Table 4.2. Outputs from the GRIDWALK model for the sloth under different SSPs (Shared Socio-economic Paths) and policy options.

Scenario and region	Number of reproductive units in viable patches	Number of total reproductive units	Proxy of cohesion	Number of networks	Number of patches in the network
current_ca	12732970.43	12759361.58	0.9979	244	14461
SSP1P_ca	12556118.28	12567267.17	0.9991	279	13884
SSP5P_ca	11586257.21	11598174.34	0.9990	307	11054
SSP5S_ca	11545280.05	11557555.58	0.9989	302	11141
current_ne	34897480.99	34963786.13	0.9981	1728	57504
SSP1P_ne	31679059.15	31760794.05	0.9974	1870	59404
SSP5P_ne	27492546.96	27572485.13	0.9971	1838	41809
SSP5S_ne	26566485.62	26660348.07	0.9965	1936	38403
current_st	15689467.14	15744456.22	0.9965	1257	33734
SSP1P_st	14261372.70	14325532.59	0.9955	1435	31622
SSP5P_st	12696760.51	12755369.26	0.9954	1319	23334
SSP5S_st	12394935.18	12456335.56	0.9951	1316	22414
current_am	109197500.03	109238821.9	0.9996	426	39519
SSP1P_am	107893611.95	107942718.6	0.9995	456	41519
SSP5P_am	103623879.53	103673394.9	0.9995	571	36719
SSP5S_am	103434360.35	103484400.8	0.9995	576	36148

The results show that for both species, in each region, current situation is better than any of the scenarios. For the future scenarios, SSP1P is best, followed by SSP5P and SSP5S is the worst. There are many more sloths than jaguars, as expected. Jaguar has a lower cohesion, with patches too small for an MVP. And in most cases, the patches are too isolated to form a network. The sloth on the other hand has high cohesion, as most patches are large enough for an MVP. It has many networks of patches, as patches are smaller and close together. The main reason for the deterioration of the Jaguar and Sloth landscape between current situation and future scenarios is that policy is assumed only to have an effect after 10 years of business as usual: under SSP5P and SSP5S the deforestation continues at the current pace, while under SSP1P the deforestation is modelled to stop after the first 10 years, leading to a larger forest cover under SSP1P than the other two scenarios. The Amazonia region has the highest numbers of both jaguars and sloths, in all scenarios.

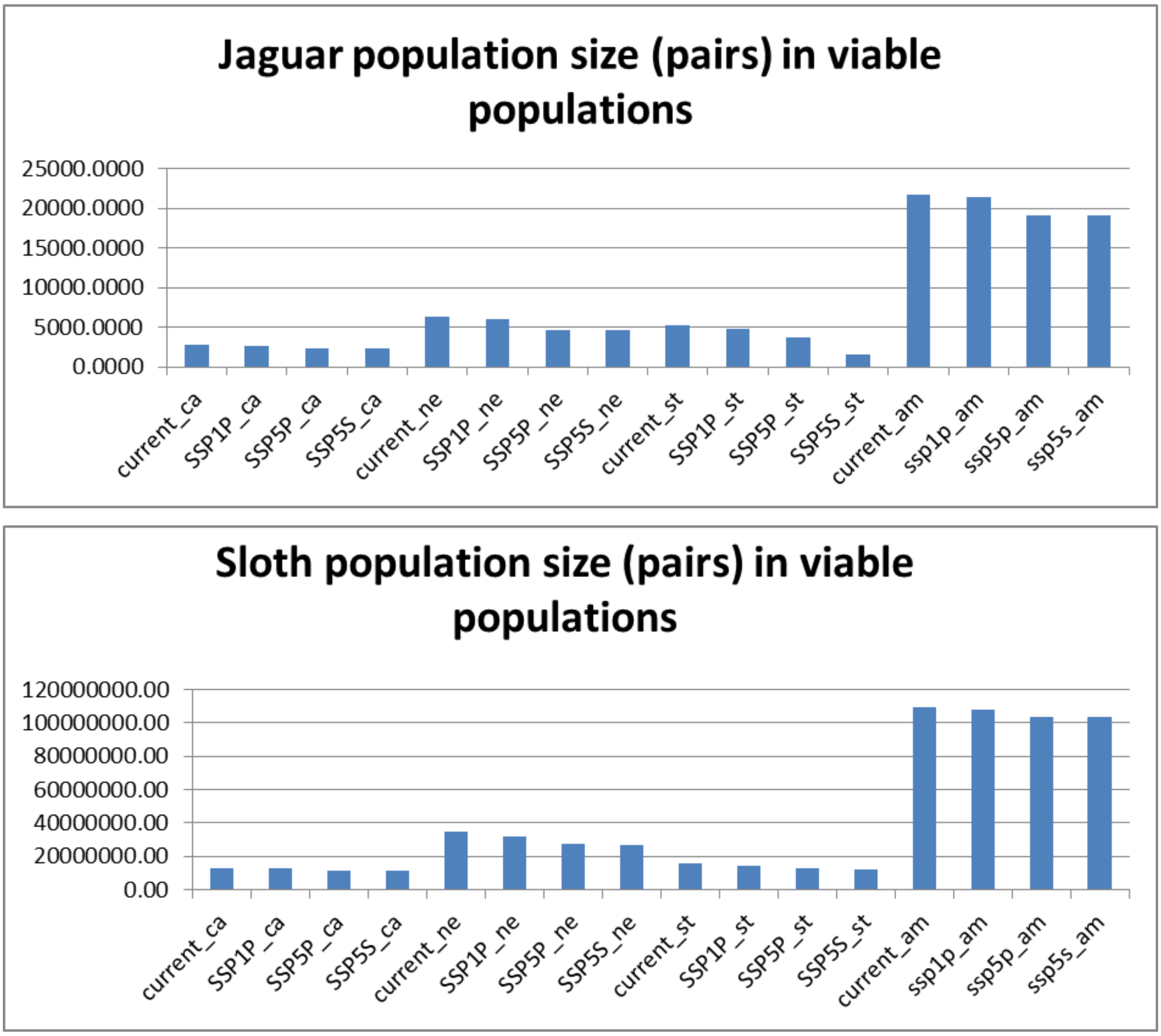


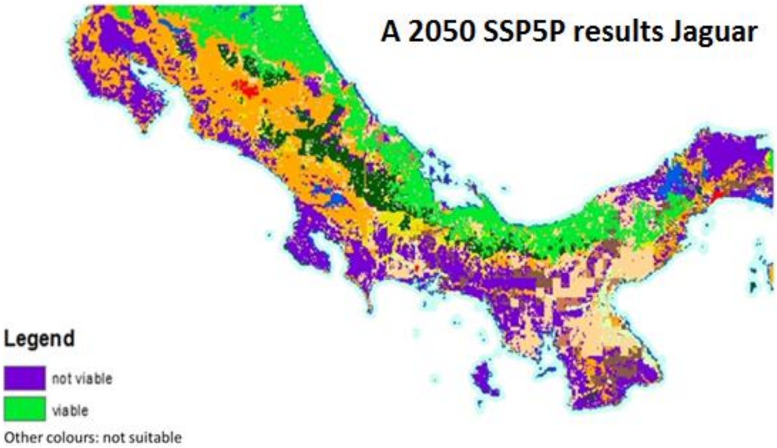
Figure 4.3a and b: the results of GRIDWALK and LARCH, showing potential population size in viable populations in the four regions (central, north-east, sub-tropical and Amazonian) for the four scenarios for jaguar (a) and sloth (b): current situation and three scenarios for 2050. See text for explanation.

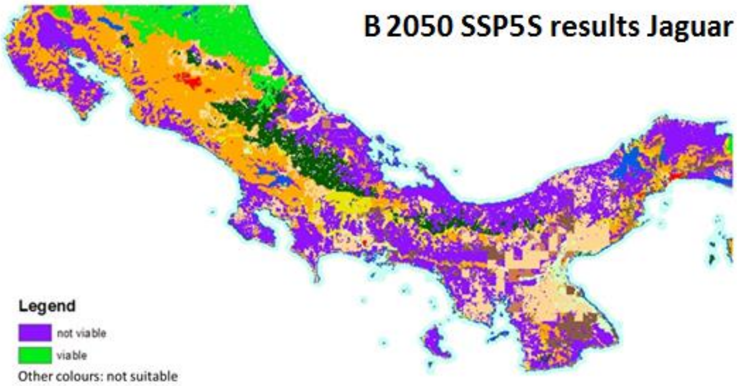
Striking are the small differences between SSP5P and SSP5S. There seems to be little effect of the policies meant for nature protection. The reason for this is the demand for agricultural products which is the same for both scenarios and drives land use change and deforestation. Under SSP5P, the forest coverage in the protected areas does not change after the first ten years but deforestation outside the protected areas continues.

It is difficult to see differences between scenarios for entire regions. We therefore zoom in and as an example give some results for Jaguar viability in Central America, a region with rather different outcomes for the three scenarios for 2050 in Figure 4. Under SSP1P, more energy crops are planted for producing biofuels, resulting in part of the current annual crop land converting to energy crop land. In order to meet the demand for annual crops, some perennial crop land is used for planting annual crops. In the model we assume that perennial crops have some value for the Jaguar, but annual crops are unsuitable. Therefore, the increase in energy crop land under SSP1P compared to SSP5P causes habitat fragmentation. In this case, although SSP1P has a better score for the entire region, locally Jaguar viability is hampered by land use changes favouring energy crops, and causing a demand driven conversion of existing perennial crops into annual crops. This causes the largest patch to be dissected into smaller, non-viable parts.

We can conclude from this connectivity analysis that future land-use change under any scenario is going to further compromise biodiversity conservation, and this will be especially the case in a business as usual scenario, and with no environmental protection policies. The only real change for the better would be is to lower the demands for crops, and to focus on other sources of energy than biofuels. An environmentally friendly future (such as SSP1P), although good for the climate, is not necessarily always better for biodiversity conservation. Locally, the focus on energy crops instead of fossil fuel can lead to extra habitat loss. Also policies meant to protect natural habitat may lead to unwanted consequences as demand-driven land use change will happen elsewhere as long as the demands remain the same.

This analysis uses models (GLOBIO and GRIDWALK/LARCH) that make many simplifying assumptions. They do not take into account climate change impacts, only land-use change. They disregard many other factors such as hunting. So the results are best used to support discussions, and are not to be regarded as a prediction of the future.





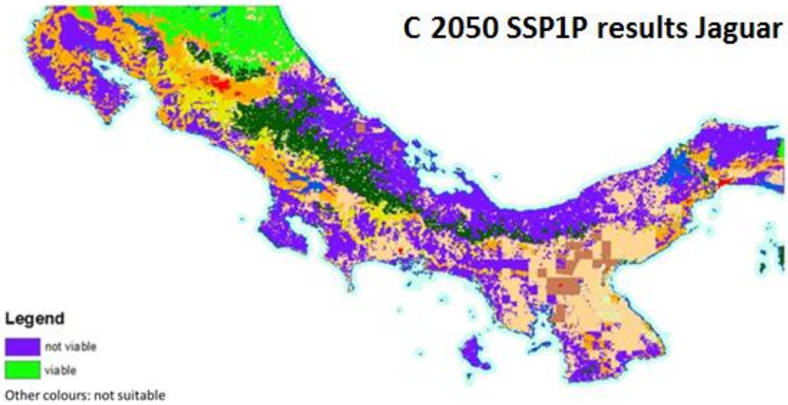


Figure 4.4. Patch viability in 2050 under the different scenarios in northern Panama and Costa Rica. (A) SSP5P, (B) SSP5S, (C) SSP1P. See text for explanation.

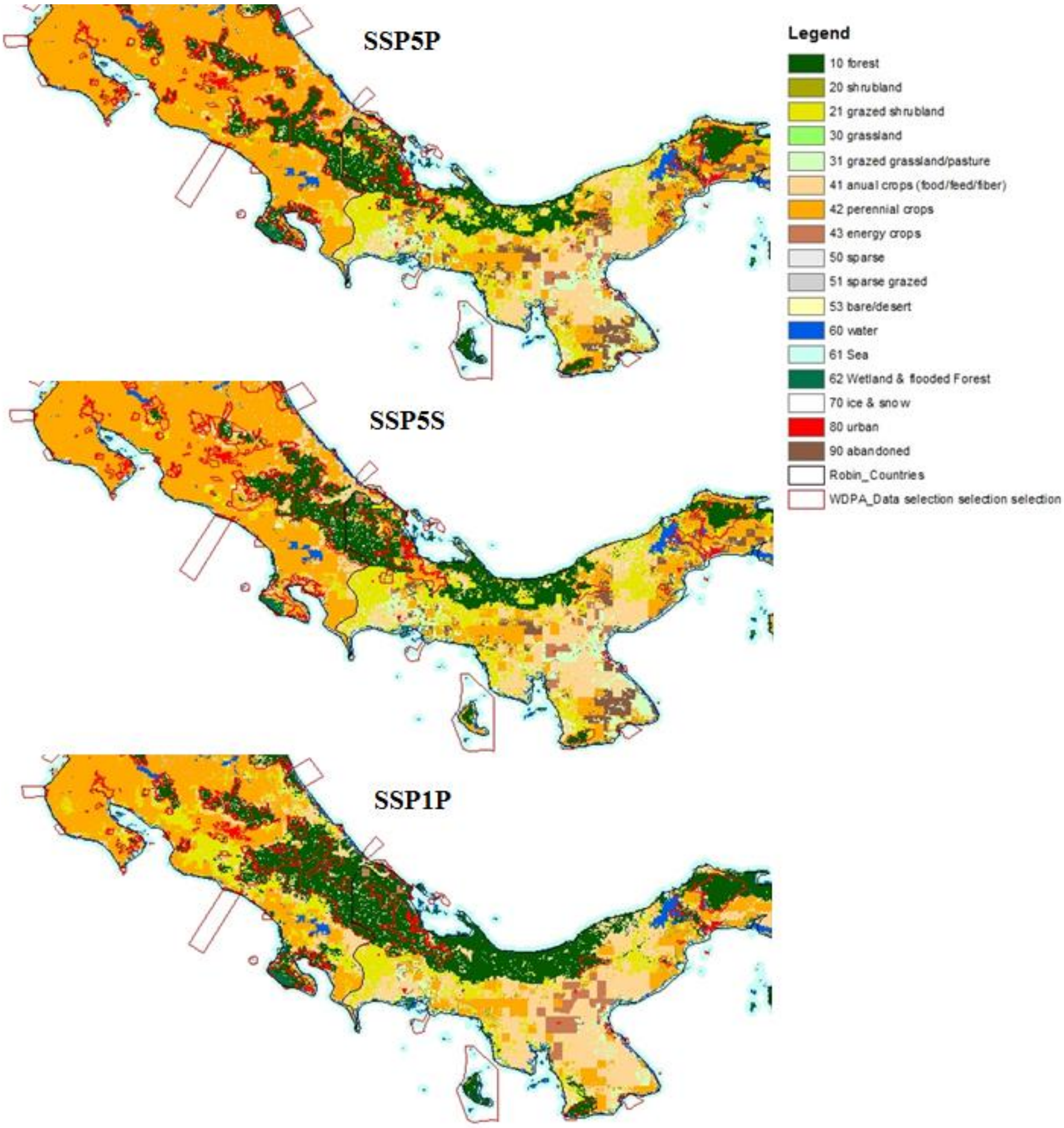


Figure 5. Land use maps of the three scenarios in Figure 4.

References:

Alkemade R, Van Oorschot M, Miles L, Nellemann C, Bakkenes M, et al. (2009) GLOBIO3: A framework to investigate options for reducing global terrestrial biodiversity loss. Ecosystems 12: 374-390.

DMA (1992) Digital Chart of the World. Fairfax, Virginia: Defense Mapping Agency.

Baveco, JM. (2010). GRIDWALK model.

Li, Y., 2015, Landscape Connectivity and Biodiversity Conservation in Latin America. WUR internship report.

Jones, L., Kok, K., (2013). D2.3.1-Scenarios for use in ROBIN. Public report D2.3.1 from the EC ROBIN project.

OECD (2012) OECD Environmental Outlook to 2050: OECD Publishing.

Opdam, P., Verboom, J. and Pouwels, R., 2003. Landscape cohesion: an index for the conservation potential of landscapes for biodiversity. Landscape ecology, 18(2), pp.113-126.

Schippers, P., Verboom, J., Knaapen, J. P., & Apeldoorn, R. V. (1996). Dispersal and habitat connectivity in complex heterogeneous landscapes: an analysis with a GIS-based random walk model. Ecography, 19(2), 97-106.

Van Eupen, M., A. Cormont, K. Kok, M. Simoes, S. Pereira, M. Kolb, and R. Ferraz. (2014). D2.2.1 Modelling land use change in Latin America. Public report D2.2.1 from the EC ROBIN project.

Verboom, J., Foppen, R., Chardon, P., Opdam, P. and Luttikhuizen, P., 2001. Introducing the key patch approach for habitat networks with persistent populations: an example for marshland birds. Biological conservation, 100(1), pp.89-101.

Verboom, J., Snep, R.P.H., Stouten, J., Pouwels, R., Pe'er, G., Goedhart, P.W., van Adrichem, M.H.C., Alkemade, J.R.M. and Jones-Walters, L.M., 2014. Using Minimum Area Requirements (MAR) for assemblages of mammal and bird species in global biodiversity assessments (No. 33, p. 22). Wettelijke Onderzoekstaken Natuur & Milieu. http://library.wur.nl/WebQuery/wurpubs/485191

5. Prevention and early warning signals.

5.1. Conclusions about the magnitude of the risk

Summarizing the above, we may conclude that there is both evidence for and against the existence of a significant risk of large scale forest dieback occurring after crossing a tipping point. The probability of it occurring in the next century is probably low, in a business-as-usual scenario with REDD+ policies implementation and a certain level of forest protection. However, the possibility of large scale dieback cannot be ruled out completely, and the consequences would be enormous. Many of the ecosystem services of tropical forests would be lost and the system could become a huge carbon source instead of a carbon sink, with important consequences not only in Latin America, but worldwide.

5.2. Prevention

Due to the feedbacks described above (Introduction), the resilience of the tropical forest ecosystems seems to depend on the persistence of large areas of pristine forests. Many pressures work in synergy to potentially cause large scale forest degradation or even dieback. Some of these processes are more difficult to stop than others. E.g., reducing global warming or influencing oscillations such as the El Niño Southern Oscillation (ENSO) is more difficult than reducing deforestation. As intact forests are crucial not only for provisioning important ecosystem services for humanity, but also for their own persistence (fire prevention, water cycling), we need to protect them. Deforestation seems (demonstrably) controllable, and depends on adequate policies. Assuming that climate change and deforestation are a deadly cocktail, we should aim at stopping deforestation. Meanwhile, the forests can help in climate change mitigation by capturing large quantities of carbon.

5.3. Transient dynamics and time lags

Our tropical forests are already changing (Van der Sande in prep.). When conditions change - due to climate change and/or land-use change - forest ecosystems will change, but the response of the ecosystems may be characterised by so-called transient dynamics, i.e. the state of the ecosystem will be a non-equilibrium state. Especially when systems approach a tipping point, it may sometimes take years, decades or centuries before a change is noticeable, and even longer before the ecosystem settles in a new state. And maybe, if conditions keep changing, the ecosystem will always be lagging behind. All change including climate change thus leads to so-called “extinction deficits” or “extinction debts” (Tilman et al, 1994), or “colonisation credits” (Nagelkerke et al. 2002, Jackson & Sax, 2010). Such time lags between environmental impacts and ecosystem response in theory buy us time to take appropriate measures to prevent detrimental effects, such as in systems with tipping points. As we will explain in the next section, early warning signals sometimes exist and can raise awareness of upcoming change (Scheffer er al., 2009). However, one should keep in mind that effects of climate change mitigation are by themselves usually also delayed. For example, changing policies, and cutting fossil fuel emissions would still be followed by global warming for many years, as temperature is lagging behind CO2 concentrations.

The role of biodiversity in forest resilience

Theoretical work on systems with tipping points shows that spatial variability in conditions smoothens the catastrophic response of the system to a changing environment (Van Nes & Scheffer 2005). It has indeed been suggested that diversity in species’ thresholds to a deteriorating environment increases the overall resilience of the system (Claussen et al. 2013). In the case of a forest ecosystem, differential responses of tree species to increasing drought may partially neutralize the threshold response of the forest to drought, even if individual species have a threshold mortality. Identifying species with low individual resilience to a deteriorating environment could function as an early warning species, like a canary in a coalmine, for drought-induced forest mortality (see below). It has been shown that the largest trees have highest sensitivity to drought (Bennett et al. 2015), suggesting such species to be a suitable indicator of critical drought.

5.4. Early warning signals

When systems approach a tipping point, the dynamics change because the resilience, the “pull” of the old stable state tends to decrease. In the ball-in-basin metaphor (see Figure 1.1) when the basin gets shallower, two things happen: (1) because of the gentler slope the ball may move further away from the lowest point after a small stochastic perturbation and (2) because of this gentler slope the return time after disturbance is longer. In this simple metaphor, an early warning signal for the collapse of the basin would be e.g. the increased variation (in ball position, due to the first mechanism described above) and the increased autocorrelation (of the position between adjacent time steps, because of the second mechanism). The former, increased amplitudes, can announce a critical point but they can also be a direct consequence of increased fluctuations in the external forcing (Boulton et al., 2013.) The latter is called “critical slowing down” in tipping point literature (Dakos et al., 2008) and is supposed to be the most important sign that resilience is deteriorating and an abrupt change (a critical transition across a tipping point) could happen. The potential for such Early Warning (EW) in real complex systems has been explored before, mainly in a theoretical context, where it has been shown mathematically that systems that contain critical transitions often show EW signs in such variability statistics on approaching the transition (Lenton, 2011; Dakos et al., 2012; Scheffer et al., 2009). There is some empirical evidence to support this, but it may take a long time to find a statistically significant effect (Lenton, 2011; Dakos et al., 2012; Scheffer et al., 2009). Of course early warning is only possible if there is a clear, measurable signal, indicating change with sufficient delay of the actual impact to allow adaptation or mitigation – taking into account that mitigation itself (e.g. ecosystem restoration) also can have delayed effects. Moreover, in the real world it could be hard to distinguish between an increasing sensitivity to random perturbations (due to a changing form of the basin) and an increasing strength of the fluctuations in external parameters (extreme events); actually, the fluctuations may even diminish as a critical point is approached (Dakos et al., 2012; Boulton et al. 2013; Meesters et al., 2015a in prep). Monitoring of autocorrelation of the system's state variables has been proposed as a tool for predicting critical transitions, as their change is a more robust predictor than increased fluctuations amplitudes (Dakos et al., 2012), though little work has been done so far on checking this hypothesis in a tropical forest stability context.

A serious challenge may be to distinguish early warning from normal fluctuations in relatively short time series. The considered monitoring data will typically have the form of a time series of annual precipitation, dry season length etc., i.e. one value per year, because sub-annual data mainly show seasonal fluctuations. We may be able to use some historical data but often these are not going back long enough or are not detailed enough. To detect a change in e.g. autocorrelation may take many decades. Boulton & Lenton found EWS for AMOC collapse, but only after 500 years of monitoring here (Boulton & Lenton, 2015). Boulton et al. (2013) investigated coupled climate-vegetation model results predicting a decline of the Amazon system with a time scale of a century, and could not detect an increase in autocorrelation time, which the authors also attributed to the too short time scale. Meesters et al. (2015a, in prep), show the same for fast decline, whereas the increase is detectable only if the decline takes a couple of centuries (See also Thomas et al., 2015).

The next section describes some of the possible indicators of change that could be used for early warning. These include independent variables, forcing factors related to the drivers and pressures of change (i.e. related to climate change and land-use change), dependent variables (the state of the forests) and more complex indicators.

5.5. Indicators of change

We are dealing with a complex socio-ecological system (see Figure 5.3) where drivers, pressures, impacts and consequences are tightly linked together through complex often non-linear relations, where many of the causal mechanisms are unknown, or only known within a small range without the possibility to extrapolate outside that range. If a change in the system occurs, it can be absorbed by the system (resilience) or lead to minor or major changes elsewhere in the system, depending on the type of feedback relations. A set of indicators was designed in order to describe and monitor the state of the system (Perez Soba et al. 2014).

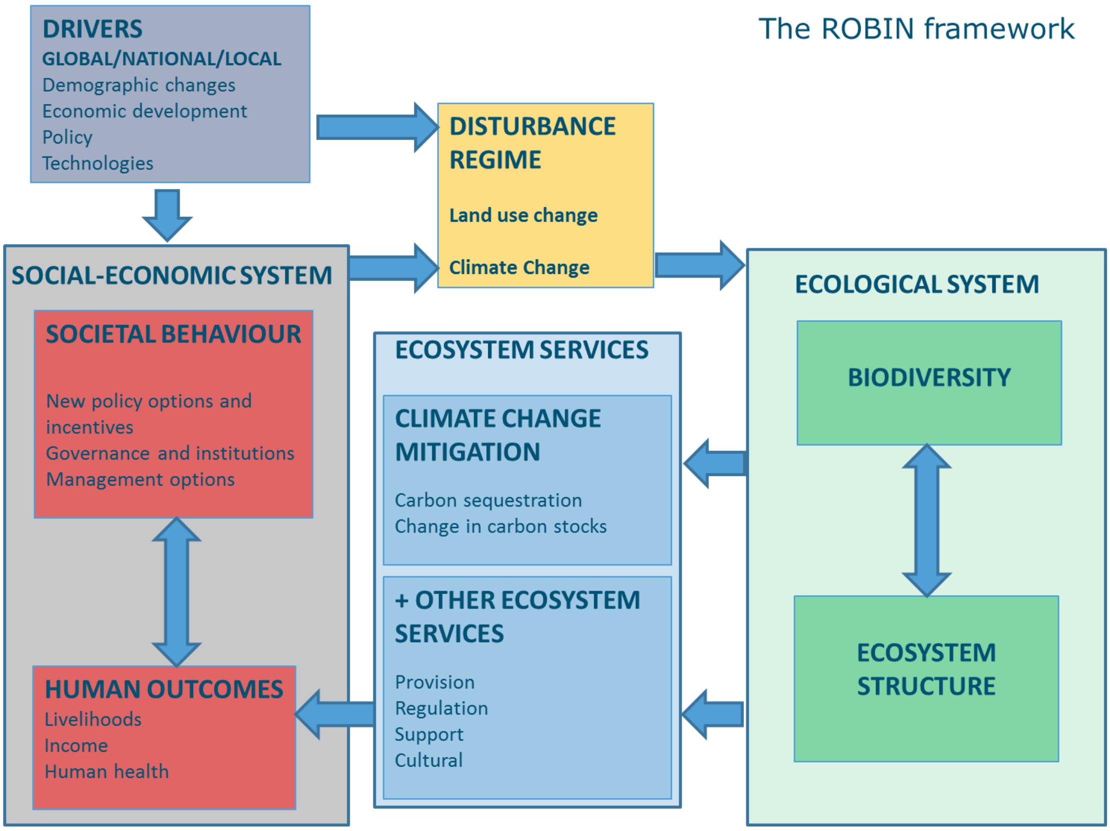


Figure 5.3. The ROBIN indicator framework which indicates the causal relations between systems.

Within the ROBIN indicator framework, many indicators can be identified that could play a role in early warning for critical transitions such as large scale forest dieback. This includes indicators from all sections: drivers, pressures (disturbance regime – we do not use the term disturbance regime in this paper because we reserve ‘disturbance’ for single system perturbations, not for ongoing pressures), the ecological system (biodiversity indicators), ecosystem services, social-economical system characteristics. We will focus here on the processes and indicators that we believe are most relevant and feasible for monitoring.

(a): indicators related to pressures (forcing factors, or 'disturbance regimes' – ROBIN framework)

In order to design sensitive indicators of change, useful for early warning, it is important to understand the different pressures, or forcing processes of change. First, we propose to distinguish three main kinds of individual pressures and associated indicators. We put indicators in bold. Please note that there may be synergy between different processes (e.g. moisture inflow and land use change).

Climate change, which is associated with global increases in CO2 concentration and air temperature, but also with decreased inflow of atmospheric moisture into South America from the Atlantic and associated changed precipitation patterns, including length of dry season in seasonal forests (Satyamurty et al., 2013, Pereira et al., 2012). NB While global and regional policies and subsequent emission reduction – or a failure to reach an agreement or to implement them – all have a major impact on the climate and can also be regarded as forcing factors, there are no straightforward SMART (specific, measurable, achievable, realistic and time-dynamic) indicators for these so we propose to focus mainly on the easily measurable (in time and space) physical indicators.

Land-use change, mainly deforestation pattern and process, which is associated with economic activity in the region, but also with several other driving factors, from local to global (e.g. global market for energy crops and biofuel – related to, amongst other things, the price of oil on the world market; meat demand and consumption in Asia). We are aware that governance factors such as regional measures to mitigate deforestation but also settlement programmes can strongly affect deforestation, as has been demonstrated for the Brazilian Amazon (Aguiar et al., 2012), and on the other hand, implementation of PES policies (payment for ecosystem services) can locally lead to reforestation (e.g. Arriagada et al., 2012; Schomers & Matzdorf 2013). Again, we propose to mainly focus on the easily measurable (by remote sensing) indicators that describe deforestation rate and pattern. As density and spatial patterns of forests and other land-use types affect evapotranspiration rates, radiation and atmospheric heat budgets, which in turn drive the hydrological cycle, it is important to record not only deforestation pattern, but also the type of land-use that has replaced the forests. Increasingly sophisticated methods are becoming available to monitor surface forest cover. For example, combinations of radar, optical and LIDAR remote sensing are enabling the observation of even very small-scale disturbance, road building and mining activities. Biomass and net primary production (NPP) should be monitored directly, preferably through remote sensing methods (visual, LIDAR and radar).

Extreme events, which are projected to increase in frequency, are expected to be important 'disturbances' that could push a system across a tipping point. Extreme events are mainly associated with extremely dry and wet years, often linked to the ENSO phenomenon but also to the North- Atlantic circulation (Marengo et al., 2008), but it is not certain that these phenomena will be the main factors causing increase in future extremes (IPCC, 2013). Extreme socio-economic events (e.g. economic or political crises) can also be important, causing migration and land-use change. Indicators for extreme events overlap with, or highly correlate with the ones already mentioned above (temperature, precipitation, loss of forest to natural disasters or climate extremes).

(b): indicators related to the state of the ecological system (response variable)

All biodiversity indicators such as number of tree species, number of plant functional types, number of animal species, ecosystem integrity indicators or even remote sensing-derived indicators such as NDVI (greenness) tend to be highly correlated and can serve as indicators of resilience (see introduction). Even simple indicators related to the state of the ecological system, which may be easy to monitor by remote sensing and are proxies of biodiversity, can be of use here such as % of forest cover, greenness (NDVI), biomass proxies (e.g. see chapter 3). Soil moisture is believed to be a crucial variable in complex forest ecosystems as it relates to tree vitality (see below).

Drought exclusion studies (Da Costa et al. 2010; Nepstad et al., 2007) show that drought stress is likely to lead to shifts in tree species composition (favouring the more drought-resilient species). The same holds for other types of stress, e.g. climate change leads to pole-ward and/or up-hill shifts of species distributions changing community composition (e.g., Pearson & Dawson 2003). Thus shifts in composition can serve as sensitive indicators of change, if we understand the relationship between stress and species resilience. However merely counting the number of species might lead to wrong conclusions, as disturbance tends to first lead to an increase in generalist species and only after a time lag leads to a decline of original species (Tilman et al., 1994; Nagelkerke et al., 2002). This is sometimes called the 'biodiversity illusionary phenomenon', or BIP effect (Colding and Folke, 2009). So monitoring, or predicting, simply the total number of species is not sufficient. It may be better to use a representative set of policy relevant species.

Several networks of Long-term ecological (LTER) monitoring exist in the region, which, combined with the rich information from other sources can serve as a source of information on the dynamics of species composition and vitality.

(c): complex indicators

The role of fire in critical transitions in the neotropical forest ecosystems is potentially large and fire indicators (magnitude, intensity and frequency) should definitely be considered as indicators of change. Indicators of fire and fire risk should be monitored. Monitoring fire risk would mean the monitoring of soil moisture – which is also an important indicator of tree viability - as well as fuel load (dead biomass and grasses) and its exposure (openness of vegetation). It is important to further develop techniques to distinguish between agricultural maintenance fires and real forest fires. Meesters and Dolman (2015, in prep.) show in a model study that soil moisture appears central and among the most sensitive factors in forest degradation. NB there is a link between sea surface temperatures and fire in South America (Chen et al. 2013) so this could be used as early warning for fire risk.

Large-scale and high-resolution (time, space) soil moisture mapping may be the most promising monitoring option for early warning (Kruijt et al. 2014, AMAZALERT deliverable and references therein). A promising new technique is under development that would enable large-scale monitoring of soil moisture status using cosmic neutron radiation (Zreda et al., 2012). Other relatively new techniques are those related to passive microwaves, i.e., radar remote sensing. Both techniques produce information on total moisture in soil plus vegetation rather than soils only. However, for the purpose of assessing fire risk and drought stress for trees this might be more an advantage than a disadvantage (Kruijt et al. 2014, AMAZALERT deliverable).

Another complex indicator could be the return time to closed canopy after (a) logging and (b) abandoning agricultural land. If the time to recovery is growing longer, it is a sign of slowing down.

In Table 1 (from the Deliverable “A blueprint for an early warning for critical transitions system in Amazonia” of the AMAZALERT project.), we present an overview and classification of possible critical transitions, their causes, effects, indicators and possible monitoring to observe such indicators.

5.6. Monitoring critical transitions – what are we looking for?

So we are dealing with a complex system, and indicators for the state of the system will fluctuate due to stochastic effects, or real, gradual change. It might take some time before we can distinguish between the two with a certain certainty (defining a probability of e.g. 5% that we falsely detect a change when there is only fluctuation). Advanced statistical analysis is necessary to identify early warning signals. It is in the end a policy decision what risk of a critical transition we find acceptable as society. The wider consideration of societal impact also brings into play the question of societal tipping points. There is recognition that these can occur in social systems (e.g. Scheffer et al. 2009), but also some debate about whether they occur in the same manner as in ecological systems (Bentley et al. 2014). Jones et al. (2014) distinguish between different types of threshold, or societally acceptable limit: system thresholds, policy targets, legislative commitments and social thresholds.

In terms of biophysical or ecosystem responses and indicators, questions to ask for early warning systems are e.g. (1) what is the variation and the autocorrelation of the system state and (2) is variation/autocorrelation changing? Increasing variation and increasing autocorrelation are early warning signals of critical transition. And (3) what level of system change is acceptable? What are the limits for each indicator?

In general it has been proposed that forests become vulnerable to drought and fire-induced decline if soil moisture drops below about 0.75% of field capacity (Nepstad et al., 2004), and can be assumed resilient to fire as long as precipitation is more than 1.2 times potential evapotranspiration (Hirota et al., 2010). For other indicators it is more difficult to determine limits.

For social indicators, there is increasing recognition that a minimum amount of resource use (and therefore land use) is necessary to maintain basic human health and wellbeing (Leach et al. 2012), and to satisfy sustainable development goals. Data to monitor and inform these are often only available at relatively coarse scale, but for some composite indicators such as the Human Development Indicator, are available at least to Municipality level in countries such as Mexico and Brazil. Monitoring of these indicators can provide evidence of some adverse impacts on social systems as a result of some types of ecosystem damage. They may also provide information on some of the societal drivers of land use change. For example, changes in net migration may be a signal of social systems in a state of flux. High immigration rates might presage increased pressure on land or natural resource use, while high outward migration might be associated with changes in traditional management practices, or a decline in the economy or the state of natural resources. More work needs to be done to establish how useful these might be as Early Warning Systems. Indicators of resource use can indicate pressure and if thresholds are known, can anticipate either potential societal problems, or increasing economic costs required to substitute those resources. There is a threshold for water available per capita of 1200 m3 per capita (FAO 1994; Yang et al. 2003). This defines the minimum amount of water required both for domestic use and for crop growth to feed the population. Below this level, there may be direct societal consequences from insufficient food production or, at the least, greater economic costs associated with importing food. Effects on human wellbeing within a country will be highly variable, and felt most heavily by those at the margins of society.

5.7. Towards an integrated monitoring system (after Kruijt et al. 2014)

From the analysis of the most likely critical transitions in neotropical forest ecosystems, it is clear that the most urgent variables for monitoring are those relating to moisture transport and soil moisture in the basin, and those relating to the dynamics of biomass, fire, and biodiversity. Several monitoring systems an initiatives already exist especially in the Amazon region (especially forest cover, biomass, biodiversity and river discharge), while in other regions and on other aspects there are still important caveats (such as (soil) moisture, rainfall and other meteorological variables, as well as forest vitality and mortality. Socio-economic indicators are available for most countries but need to be harmonized.

The various model simulations performed in AMAZALERT indicate (as reviewed in Kruijt et al. 2014): 1) high risk of degradation in the SE of Amazonia, because of the combination of land-use change risk and climate; 2) high uncertainty about both climate change and fire susceptibility in the NE (i.e., Northern state of Para, and Guianas); 3) a robust climate and forests in the NW; and 4) a yet relatively undisturbed but very sensitive SW, which is at the same time essential for moisture recycling to the south of the continent. This suggests that more intensive monitoring is especially needed in the NE (particularly N Para state) and the SW (particularly S. Amazonas state, Acre, S. Peru and N Bolivia). Rainfall monitoring should be intensified also in the NW, whereas in the SE, a strong focus should be on shorter-term forecasting and adaptation to change.

5.8. How to prevent or reverse critical transitions when early warning signals indicate a high risk

When conditions have changed due to human-induced forcing factors – e.g. a combination of land-use change and climate change-, or natural factors, and we assume that a large scale forest dieback is happening or is likely to happen (the likelihood we find acceptable is a matter of policy decisions), what can be done? If a complex system has alternative stable states, it is not enough to restore the conditions. And that might be even very difficult as climate change is going to continue for a long time even if we would abolish the use of fossil fuels tomorrow. If we stop deforestation tomorrow and start reforestation projects, it might take decades or centuries before ecosystems are restored (Cole et al. 2014). Still there are a few things that can be done:

Halting deforestation and even reversing the trend (reforestation) is easier than stopping climate change, and this could restore hydrological cycling where it has deteriorated

Mitigating climate change by safeguarding biodiversity, which helps to insure long-term carbon storage and sequestration in tropical forests (Poorter et al., 2015)

Large scale change from annual to permanent crops would help preserve the water cycle because of the evapotranspiration properties of permanent crops which are closer to forest than the properties of annual crops.

Restoring ecosystem services such as pollination and seed dispersal by re-introduction of previously (in the near past) extinct species might aid ecosystem restoration as well as service provision to humanity

Extreme events such as extremely wet years can be used to help systems to tip back from degraded low biomass low biodiversity dry systems to restored high biomass high biodiversity high water cycling systems - by focussing on reforestation of degraded lands in wet years, in dry areas where water availability has become a limiting factor (Holmgren et al. 2001; Holmgren et al. 2013.

6. References

Aguiar, A. P. D., Ometto, J. P., Nobre, C., Lapola, D. M., Almeida, C., Vieira, I. C., Soares, J. V., Alvala, R., Saatchi, S., Valeriano, D. & Castilla-Rubio, J. C. (2012) Modeling the spatial and temporal heterogeneity of deforestation-driven carbon emissions: the INPE-EM framework applied to the Brazilian Amazon Global Change Biology 18(11), 3346-3366.

Aragão, L. E. O. C., 2012. Environmental science: the rainforest’s water pump. Nature, 489, 217-218, doi: 10.1038/nature11485

Arriagada, R.A., Ferraro, P.J., Sills, E.O., Pattanayak, S.K. and Cordero-Sancho, S., 2012. Do payments for environmental services affect forest cover? A farm-level evaluation from Costa Rica. Land Economics, 88(2), pp.382-399.

Avissar, Roni, and David Werth. "Global hydroclimatological teleconnections resulting from tropical deforestation." Journal of Hydrometeorology 6.2 (2005): 134-145.

Baudena et al. 2015, Forests, savannas, and grasslands: bridging the knowledge gap between ecology and Dynamic Global Vegetation Models, Biogeosciences 12, 1833-1848.

Bennett et al. 2015, Larger trees suffer most during drought worldwide. Nature Plants 1, article no 15139.

Bentley, R. A., Maddison, E. J., Ranner, P. H., Bissell, J. J., Caiado, C. C. C. S., Bhatanacharoen, P., Clark, T., Botha, M., Akinbami, F., Hollow, M., Michie, R., Huntley, B., Curtis, S. E. and Garnett, P. R. (2014) 'Social tipping points and Earth systems dynamics.', Frontiers in environmental science. Vol 2 . p. 35.

Boulton, C.A., Good, P., and Lenton, T.M., 2013. Early warning signals of simulated Amazon rainforest dieback. Theoretical Ecology. DOI 10.1007/s12080-013-0191-7.

Boulton, C. A., & Lenton, T. M. (2015). Slowing down of North Pacific climate variability and its implications for abrupt ecosystem change. Proceedings of the National Academy of Sciences, 112(37), 11496-11501.)

Chen et al. 2011, Forecasting Fire Season Severity in South America Using Sea Surface Temperature Anomalies, Science 334, 787-791.

Chen et al. 2013, Satellite observations of terrestrial water storage provide early warning information about drought and fire season severity in the Amazon, JOURNAL OF GEOPHYSICAL RESEARCH: BIOGEOSCIENCES, VOL. 118, 1–10.

Claussen et al. 2013, Simulated climate-vegetation interaction in semi-arid regions affected by plant diversity. Nature Geoscience 6, 954-958.

Colding, J., Folke, C., 2009. The role of golf courses in biodiversity conservation and ecosystem management. Ecosystems 12, 191-206.

Cole, L. E., Bhagwat, S. A., & Willis, K. J. (2014). Recovery and resilience of tropical forests after disturbance. Nature communications, 5.

Cox, P.M., Betts, R.A., Jones, C.D., Spall, C.A. and Totterdell, I.J., 2000. Acceleration of global warming due to carbon-cycle feedbacks in a coupled climate model. Nature 408: 184-187.

Cox et al (2004) Amazonian forest dieback under climate-carbon cycle projections for the 21st century, Theoretical and Applied Climatology, 78

da Costa, A.C.L., Galbraith, D., Almeida, S., Takeshi Tanaka Portela, B., da Costa, M., de Athaydes Silva Junior, J., Braga, A.P., de Gonçalves, P.H.L., de Oliveira, A.A.R., Fisher, R., Phillips, O.L., Metcalfe, D.B., Levy, P and Meir, P., 2010. Effect of 7 yr of experimental drought on vegetation dynamics and biomass storage of an eastern Amazonian rainforest. New Phytologist, 187(3): 579-591. Doi: 10.1111/j.1469-8137.2010.03309.x.

Dakos, V., Scheffer, M., van Nes, E. H., Brovkin, V., Petoukhov, V., & Held, H. (2008). Slowing down as an early warning signal for abrupt climate change. Proceedings of the National Academy of Sciences, 105(38), 14308-14312.

Dakos, V., van Nes, E.H., d'Odorico, P. and Scheffer, M., 2012. Robustness of variance and autocorrelation as indicators of critical slowing down. Ecology 93: 264-271.

Davidson, E.A., de Araujo, A.C., Artaxo, P., Balch, J.K., Brown, I.F., Bustamante, M.M.C., Coe, M.T., DeFries, R.S., Keller, M., Longo, M., Munger, J.W., Schroeder, W., Soares-Filho, B.S., Souza Jr, C.M. and Wofsy, S.C., 2012. The Amazon basin in transition. Nature 481: 321–328. Doi:10.1038/nature10717.

Díaz, S., Fargione, J., Chapin, F. S., & Tilman, D. (2006). Biodiversity loss threatens human well-being. PLoS biology, 4(8), 1300-1305.

Doughty, Wolf & Malhi 2013, The legacy of the Pleistocene megafauna extinctions on nutrient availability in Amazonia, Nature Geoscience 6, 761-764

Elmqvist, T., Folke, C., Nyström, M., Peterson, G., Bengtsson, J., Walker, B., & Norberg, J. (2003). Response diversity, ecosystem change, and resilience. Frontiers in Ecology and the Environment, 1(9), 488-494.

FAO, 1994. Water for Life. World Food Day 1994, Rome

Folke, C., Colding, J., & Berkes, F. (2003). Synthesis: building resilience and adaptive capacity in social-ecological systems. Navigating social-ecological systems: Building resilience for complexity and change, 352-387.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., & Holling, C. S. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. Annual Review of Ecology, Evolution, and Systematics, 557-581.

Galbraith, D., Levy, P.E., Sitch, S., Huntingford, C., Cox, P., Williams, M., and Meir, P., 2010. Multiple mechanisms of Amazonian forest biomass losses in three dynamic global vegetation models under climate change. New Phytologist, 187(3): 647-65. Doi: 10.1111/j.1469-8137.2010.03350.x.

Gatti, L.V., Gloor, M., Miller, J.B., Doughty, C.E., Mahli, Y., Domingues, L.G., Basso, L.S., Martimewski, A., Correia, C.S.C., Borges, V.F., Freitas, S., Braz, R., Anderson, L.O., Rocha, H., Grace, J., Phillips, O.L. and Lloyd, J., 2014. Drought sensitivity of Amazonian carbon balance revealed by atmospheric measurements. Nature, 506: 76–80. Doi:10.1038/nature12957.

Global Carbon Project, 2014. Carbon budget and trends 2014 www.globalcarbonproject.org/carbonbudget

Good, P., Jones, C.D., Lowe, J., Betts, R.A., Booth, B.B.B., and Huntingford, C., 2011. Quantifying environmental drivers of future tropical forest extent. J. Climate 24: 1337-1349.

Good, P., Jones, C.D., Lowe, J., Betts, R.A., and Gedney, N., 2013. Comparing tropical forest projections from two generations of Hadley Centre Earth System Models, HadGEM2-ES and HadCM3LC. J. Climate 26(2): 495-511.

Hirota, M., Holmgren, M., Van Nes, E.H. and Scheffer, M., 2011. Global resilience of tropical forest and savanna to critical transitions. Science 334(6053):232–235. Doi:10.1126/science.12106.

Hoffmann et al. 2012, Ecological thresholds at the savanna-forest boundary: how plant traits, resources and fire govern the distribution of tropical biomes, Ecology Letters 15, 759-768.

Holmgren, M., & Scheffer, M. (2001). El Niño as a window of opportunity for the restoration of degraded arid ecosystems. Ecosystems, 4(2), 151-159.

Holmgren, M., Hirota, M., van Nes, E.H. and Scheffer, M., 2013, Effects of interannual climate variability on tropical tree cover. Nature Climate Change, 3: 755–758. Doi:10.1038/nclimate1906.

Huntingford, C., Zelazowski, P. , Galbraith, D., Mercado, L.M., Sitch, S., Fisher, R., Lomas, M., Walker, A.P., Jones, C.D., Booth, B.B.B., Malhi, Y., Hemming, D., Kay, G., Good, P., Lewis, S.L., Phillips, O.L., Atkin, O.K., Lloyd, J., Gloor, E., Zaragoza-Castells, J., Meir, P., Betts, R., Harris, P.P., Nobre, C., Marengo, J., and Cox, P.M., 2013. Simulated resilience of tropical rainforests to CO2-induced climate change. Nature Geoscience 6, 268-273. Doi 10.1038/NGEO1741.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jones L., Winograd, M., Perez-Soba, M., Balvanera, P., Lazos, E., Verboom, J., Baveco, H. (2014). Definition of regional sustainability limits. EC public report on D2.1.3 of the ROBIN project.

Kay, G., Alves, L., Boisier, J.P., Boorman, P., Boulton, C., Ciais, P., Cardoso, M., Good, P., Hemming, D., Marengo, J., Meesters, A. and Sampaio, G., 2014. Report on estimated likelihood for irreversible collapse. Deliverable D3.4, Amazalert.

Kramer, K., Groen, T. A., & van Wieren, S. V. (2003). The interacting effects of ungulates and fire on forest dynamics: an analysis using the model FORSPACE. Forest Ecology and Management, 181(1), 205-222.

Kruijt, B., Jans, W.W.P., Vasconcelos, S., Almeida Melém, E., Tribuzy, E.S., Felsembourgh, C., Rowland, L., Lola da Costa, A., Meir, P., Chambers, J. and Camargo, P., 2015. Temperature response of leaf photosynthesis of Amazon forest trees and its acclimation to mean temperature change. In Preparation.

Lapola, D.M., Oyama, D.M., Nobre, C.A., 2009. Exploring the range of climate biome projections for tropical South America: the role of CO2 fertilization and seasonality. Global Biogeochemical Cycles 23: GB300.

Laundré, J. W., Hernández, L., & Altendorf, K. B. (2001). Wolves, elk, and bison: reestablishing the" landscape of fear" in Yellowstone National Park, USA. Canadian Journal of Zoology, 79(8), 1401-1409.

Leach, M., Rokstrom, J., Raskin, P., Scoones, I. C., Stirling, A. C., Smith, A. ... & Olsson, P. (2012). Transforming innovation for sustainability. Ecology and Society, 17(2), 11.

Lenton, T.M., Held, H., Kriegler, E., Hall, J., Lucht, W., Rahmstorf, S. and Schellnhuber, H.J., 2008. Tipping elements in the Earth's climate. Proceedings of the National Academy of Sciences of the United States of America,105(6): 1786-1793. Doi: 10.1073/pnas.0705414105

Lenton, T.M., 2011. Early warning of climate tipping points. Nature Climate Change 1:201–209. Doi:10.1038/nclimate1143.

Lewis SL, Brando PM, Phillips OL, van der Heijden GMF, Nepstad D (2011) The 2010 Amazon drought. Science 331:554. doi: 10.1126/science.1200807.

Malhi et al (2009) Exploring the likelihood and mechanism of a climate-change-induced dieback of the Amazon rainforest, PNAS 106(49).

Marengo, J. A., 2004. Interdecadal variability and trends of rainfall across the Amazon basin. Theor. Appl. Climatol. 78: 79–96.

Marengo, J.A., Nobre, C. A., Tomasella, J., Oyama, M. D., Sampaio, G., Oliveira, R., Camargo, H., Alves, L. M., Foster Brown, I., 2008: The Drought of Amazonia in 2005. J. Climate, 21, 495–516. doi: http://dx.doi.org/10.1175/2007JCLI1600.1

Martorano, L.G.; Perreira, L.C.; César, E.G.M.; Pereira, I.C.B. 1993. Estudos climáticos do Estado do Pará, classificação climática (Köppen) e deficiência hídrica (Thornthwhite Mather). Belém: SUDAM/EMBRAPA, SNLCS, 1993. 5p.

Meesters, A.G.C.A., Good, P., and Dolman, A.J., 2015a, in prep. Perspectives for early warning from fluctuations, illustrated with simple models for forest dieback.

Meesters, A.G.C.A. and Dolman, A.J., 2015b, in prep. Stability of the Amazon rainforest investigated with a zero-dimensional model.

Meir, P. and Woodward ??, 2010. Amazonian rain forests and drought: response and vulnerability. New Phytologist, 187: 553–557.

Mittermeier, R. A., Mittermeier, C. G., Brooks, T. M., Pilgrim, J. D., Konstant, W. R., Da Fonseca, G. A., & Kormos, C. (2003). Wilderness and biodiversity conservation. Proceedings of the National Academy of Sciences, 100(18), 10309-10313.

Nagelkerke, K. C., Verboom, J., Van Den Bosch, F., & Van De Wolfshaar, K. (2002). Time lags in metapopulation responses to landscape change. In Applying landscape ecology in biological conservation (pp. 330-354). Springer New York.

Nepstad, D., Lefebvre, P., Lopes da Silva, U., Tomasella, J., Schlesinger, P., Solórzano, L., Moutinho, P., Ray, D. and Guerreira Benito, H., 2004. Amazon drought and its implications for forest flammability and tree growth: a basin-wide analysis. Global Change Biology 10: 704-717.

Nepstad, D.C., Tohver, I.M., Ray, D., Moutinho, P. and Cardinot. G., 2007. Mortality of large trees and lianas following experimental drought in an Amazon forest. Ecology 88: 2259-2269.

Nobre, C.A., Borma, L.D., 2009. 'Tipping points' for the Amazon forest. Current opinion in environmental sustainability, 1(1): 28-36.

Oyama, M.D., Nobre, C.A., 2003. A new climate-vegetation equilibrium state for Tropical South America. Geophys Res Lett, 30(23): 2199.

Pearson, R. G., & Dawson, T. P. (2003). Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful?. Global ecology and biogeography, 12(5), 361-371.

Pereira, M. P. S., A. C. M. Malhado, and M. H. Costa (2012), Predicting land cover changes in the Amazon rainforest: An ocean-atmosphere-biosphere problem, Geophys. Res. Lett., 39, L09713, doi:10.1029/2012GL051556.

Poorter, L., van der Sande, M. T., Thompson, J., Arets, E. J. M. M., Alarcón, A., Álvarez-Sánchez, J., Ascarrunz, N., Balvanera, P., Barajas-Guzmán, G., Boit, A., Bongers, F., Carvalho, F. A., Casanoves, F., Cornejo-Tenorio, G., Costa, F. R. C., de Castilho, C. V., Duivenvoorden, J. F., Dutrieux, L. P., Enquist, B. J., Fernández-Méndez, F., Finegan, B., Gormley, L. H. L., Healey, J. R., Hoosbeek, M. R., Ibarra-Manríquez, G., Junqueira, A. B., Levis, C., Licona, J. C., Lisboa, L. S., Magnusson, W. E., Martínez-Ramos, M., Martínez-Yrizar, A., Martorano, L. G., Maskell, L. C., Mazzei, L., Meave, J. A., Mora, F., Muñoz, R., Nytch, C., Pansonato, M. P., Parr, T. W., Paz, H., Pérez-García, E. A., Rentería, L. Y., Rodríguez-Velazquez, J., Rozendaal, D. M. A., Ruschel, A. R., Sakschewski, B., Salgado-Negret, B., Schietti, J., Simões, M., Sinclair, F. L., Souza, P. F., Souza, F. C., Stropp, J., ter Steege, H., Swenson, N. G., Thonicke, K., Toledo, M., Uriarte, M., van der Hout, P., Walker, P., Zamora, N. and Peña-Claros, M. (2015). Diversity enhances carbon storage in tropical forests. Global Ecology and Biogeography. [Website | DOI: http://dx.doi.org/10.1111/geb.12364]

Rammig A, Jupp T, Thonicke K, Tietjen B, Heinke J, Ostberg S, Lucht W, Cramer W, Cox P (2010) Estimating the risk of Amazonian forest dieback. New Phytol 187:694–706. doi: 10.1111/j.1469-8137.2010.03318.

Sampaio, G., Nobre, C., Costa, M.H., Satyamurty, P., Soares-Filho, B.S. and Cardoso, M., 2007. Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. Geophys. Res. Lett., 34, L17709. Doi:10.1029/2007GL030612.

Satyamurty, P., Costa , C.P. W., Manzi, A. O., 2013. Moisture source for the Amazon Basin: a study of contrasting years, Theoretical and Applied Climatology, 111, 195-209, DOI 10.1007/s00704-012-0637-7

Scheffer, M., Bacompte, J., Brock, W.A., Brovkin, V., Carpenter, S.R., Dakos, V., Held, H., van Nes, E.H., Rietkerk, M., and Sugihara, G., 2009. Early warning signals for critical transitions. Nature 461: 53-59.

Scheffer, M. and Jeppesen, E., 2007. Regime shifts in shallow lakes. Ecosystems, 10(1), pp.1-3.

Schippers, P., van Teeffelen, A. J., Verboom, J., Vos, C. C., Kramer, K., & WallisDeVries, M. F. (2014). The impact of large herbivores on woodland–grassland dynamics in fragmented landscapes: The role of spatial configuration and disturbance. Ecological Complexity, 17, 20-31.

Schomers, S. and Matzdorf, B., 2013. Payments for ecosystem services: a review and comparison of developing and industrialized countries. Ecosystem Services, 6, pp.16-30.

Silvestrini RA, Soares-Filho BS, Nepstad D, Coe M, Rodrigues H, Assunção R (2011) Simulating fire regimes in the Amazon in response to climate change and deforestation. Ecol Appl a Publ Ecol Soc Am 21:1573–90.

Spraklen, D. V., S. R. Arnold, and C. M. Taylor. "Observations of increased tropical rainfall preceded by air passage over forests." Nature 489.7415 (2012): 282-285

Staal, A., Dekker, S. C., Hirota, M., & van Nes, E. H. (2015). Synergistic effects of drought and deforestation on the resilience of the south-eastern Amazon rainforest. Ecological Complexity, 22, 65-75.

Staver, A.C., Archibald, S., Levin, S.A., 2011. The global extent and determinants of savanna and forest as alternative biome states. Science 334(6053): 230–232. Doi:10.1126/science.1210465.

ter Steege, H. N., et al., 2006. Continental-scale patterns of canopy tree composition and function across Amazonia. Nature, 443: 444–447. DOI: 10.1126/science.1243092.

Thomas, Z. A., Kwasniok, F., Boulton, C. A., Cox, P. M., Jones, R. T., Lenton, T. M., & Turney, C. S. M. (2015). Early warnings and missed alarms for abrupt monsoon transitions. Climate of the Past Discussions, 11(2).

Tilman, D., May, R. M., Lehman, C. L., & Nowak, M. A. (1994). Habitat destruction and the extinction debt. Nature 371, 65 – 66.

Travis, J. M. J. (2003). Climate change and habitat destruction: a deadly anthropogenic cocktail. Proceedings of the Royal Society of London B: Biological Sciences, 270(1514), 467-473.

Van Nes, E.H., Hirota, M., Hollmgren, M. and Scheffer, M., 2014. Tipping points in tropical tree cover: linking theory to data. Global Change Biology, 20(3): 1016–1021. Doi: 10.1111/gcb.12398.

Van Nes & Scheffer 2005, Implications of spatial heterogeneity for catastrophic regime shifts in ecosystems. Ecology 86, 1797-1807

Verboom, J., & Wamelink, W. (2005). Spatial modeling in landscape ecology. In Issues and perspectives in landscape ecology (pp. 79-89). Cambridge University Press.

White, A., Cannell, M.G.R. and Friend, A.G., 2000. CO2 stabilization, climate change and the terrestrial carbon sink. Global Change Biol., 6: 817-833. Doi:10.1046/j.1365-2486.2000.00358.x.

Yang, H, Reichert P, Abbaspour KC, and Zehnder AJB. 2003. "A water resources threshold and its implications for food security." Environmental Science & Technology (American Chemical Society) 37: 3048-3054.

Zemp, DC, Schleussner C-F, Barbosa HMJ, van der Ent RJ, Donges JF, Heinke J, Sampaio G, Rammig A (2014) On the importance of cascading moisture recycling in South America. Atmos. Chem. Phys. 14, 13337–13359

Zreda, M., Shuttleworth, W.J., Xeng, X., Zweck, C., Desilets, D., Franz, T.E. and Rosolem, R., 2012. COSMOS: The COsmic-ray Soil Moisture Observing System, Hydrol. Earth Syst. Sci., 16: 4079–4099. Doi:10.5194/hess-16-1-2012.

Appendix 1. A Toy Model of Effect of Land Use Change on Rainfall Recycling and Downstream Crop Viability

A Toy Model of Effect of Land Use Change on Rainfall Recycling and Downstream Crop Viability

Chris Huntingford and co

August 6, 2015

1 Governing Equations

1.1 Atmospheric Considerations

We consider a column of air above the land surface, of height H (m), moving with mean velocity $u(x,t)$ (m s^{-1}) from east to west. Quantity x (km) is distance from Atlantic Coast, and the column of air has a vertically-invariant humidity $q(x,t)$ (kg water (kg air)$^{-1}$). Boundary conditions are precipitation from the column $P(x,t)$ (mm day^{-1}) and evaporation from the land surface in to the box $R(x,t)$ (mm day^{-1}).

Let the column of air be in range $x_o \leq x \leq x_o + \delta x$, it have a meridonal distance δy. Let ρ (kg air m^{-3} air) be atmospheric density. Hence the total water in the column is $q(x,t)\rho H \delta x \delta y$. For an invariant windspeed, then in a short period of time δt, the change in column water satisfies:

$$\underbrace{\rho H\left[q(x,t+\delta t)-q(x,t)\right]\delta x\delta y}_{\text{Change in water store}} = \underbrace{\delta t\delta y H u\rho q(x_o)}_{\text{Water from east}} - \underbrace{\delta t\delta y H u\rho q(x_o+\delta x)}_{\text{Water to west}} - \underbrace{\delta t\delta x\delta y\frac{P}{86400}\frac{\rho_{water}}{1000}}_{\text{Rainfall out}} + \underbrace{\delta t\delta x\delta y\frac{R}{86400}\frac{\rho_{water}}{1000}}_{\text{Evaporation in}} \quad (1)$$

and where ρ_{water} (kg m^{-3}) is density of water. Taking $\rho_{water} \approx 1000$, and performing Taylor series expansions as $q(x,t+\delta t) = q(x) + (\partial q/\partial t)\delta t$ and $q(x+\delta x,t) = q(x) + (\partial q/\partial x)\delta x$, then this gives equation:

$$\rho H\frac{\partial q}{\partial t} = -\rho H u\frac{\partial q}{\partial x} - \frac{(P(x,t)-R(x,t))}{86400}\times 0.001. \quad (2)$$

Equations are needed for P and R to close the system. At its simplest level, P is assummed linear by co-efficient μ (mm day^{-1}) (kg kg^{-1})$^{-1}$, whereby

$$P = \mu q. \quad (3)$$

1.2 Evaporation Considerations

The equation for rainfall re-cycling is given as dependent on a simple function of soil moisture availability, and additionally based on root depth of vegetation. We assume that vegetation can transpire at a rate based on the mean (by depth) soil moisture $\theta(x,t)$ (m^3 water (m^3 soil)$^{-1}$) in the root zone. Vegetation is assumed to freely transpire at a maximum rate R_{max} if soil moisture is between saturation θ_{sat} and the critical moisture content θ_{crit}, and decreases linearly with gradient β as soil moisture approaches the wilting point θ_{wilt}. Hence:

$$R(x,t) = R_{max} \qquad \text{if} \quad \theta_{sat} \geq \theta \geq \theta_{\text{crit}} \quad (4)$$

$$R(x,t) = \beta R_{max} = \frac{\theta-\theta_{wilt}}{\theta_{sat}-\theta_{wilt}} \quad \text{if} \quad \theta_{crit} \geq \theta \geq \theta_{wilt} \quad (5)$$

If vegetation has a root depth of r_d (m) then available water for potential evaporative recycling (in units of kg m^{-2}) is:

$$r_d\left(\theta-\theta_{wilt}\right). \quad (6)$$

Changes in the quantity above are forced by the rainfall and recycling fluxes, and therefore

$$r_d\frac{\mathrm{d}\theta(x,t)}{\mathrm{d}t} = \frac{(P(x,t)-R(x.t))}{86400}. \quad (7)$$

In the event that $\theta = \theta_{sat}$ and $P > R$, then excess water, $P - R$ is assummed to either move laterally to rivers, or in large differences of P minus R, then cause flooding / wetland formation.

1

1.3 Land use, simulation framework and boundary conditions

The purpose of this simplistic model is to understand the effects of any intermediate vegetation type between deep-rooted rainforest and inland crop viability. A threshold for crop viability is given by parameter β^*, and if a time occurs at which $\beta < \beta^*$ then a crop failure is deemed as having occurred. We consider spatial boundary conditions for two scenarios. In the first, we have a high rooting depth $r_{d,tree}$ (m) corresponding to tree-like vegetation propagating inland from the coast to a boundary where crops are grown, the latter with a much smaller depth $r_{d,crop}$ (m). Then we consider a second scenario with an intermediate degraded vegetation through earlier land use, called shrub, and given by $r_{d,shrub}$ (m). Here $r_{d,tree} > r_{d,shrub} > r_{d,grass}$. In scenario one, trees cover distance L_{tree} (km), crops cover distance L_{crop} (km), and in scenario two, shrubs cover a distance L_{shrub}, and where in this scenarios, $L_{tree} + L_{shrub}$ equal L_{tree} in scenario one. Hence:

$$r_d = r_{d,tree} \quad \text{for} \quad 0 \le x < 1000L_{tree} \quad \text{Scenario 1} \tag{8}$$
$$r_d = r_{d,crop} \quad \text{for} \quad 1000L_{tree} \le x < 1000(L_{tree} + L_{crop}) \quad \text{Scenario 1} \tag{9}$$
$$r_d = r_{d,tree} \quad \text{for} \quad 0 \le x < 1000L_{tree} \quad \text{Scenario 2} \tag{10}$$
$$r_d = r_{d,shrub} \quad \text{for} \quad 1000L_{tree} \le x < 1000(L_{tree} + L_{shrub}) \quad \text{Scenario 2} \tag{11}$$
$$r_d = r_{d,crop} \quad \text{for} \quad 1000(L_{tree} + L_{shrub}) \le x < 1000(L_{tree} + L_{shrub} + L_{crop}) \quad \text{Scenario 2} \tag{12}$$
$$\tag{13}$$

Temporal boundary conditions are given by specification of $q(0,t)$, i.e. at the coast-land interface. This is the seasonal driver of our modelling framework, and takes two values for wet season $q(0,t) = q_{wet}$ (kg kg^{-1}) and for dry season $q(0,t) = q_{dry}$ (kg kg^{-1}). Hence to a time τ (day) that marks the transition from wet to dry season, then:

$$q(0,t) = q_{wet} \quad \text{for} \quad 0 \le 86400t < \tau \tag{14}$$
$$q(0,t) = q_{dry} \quad \text{for} \quad \tau \le 86400t < 365 \times 86400 \tag{15}$$
$$\tag{16}$$

1.4 Model parameters

Illustrative parameters are set for our model, as follows:

Quantity	Variable Name	Units	Value
Atmospheric Height	H	m	5000
Wind speed	u	m s^{-1}	2.5
Atmospheric Density (approximated at ground level)	ρ	kg m^{-3}	1.26
Rainfall parameter	μ	mm day^{-1} (kg kg^{-1})$^{-1}$	6/0.01
Maximum evaporative flux	R_{max}	mm day^{-1}	3
Soil moisture saturation	θ_{sat}	m^3 m^{-3}	0.456
Soil moisture critical point	θ_{crit}	m^3 m^{-3}	0.31
Soil moisture wilting point	θ_{wilt}	m^3 m^{-3}	0.221
Tree coverage (Scenario 1)	L_{tree}	km	1000
Tree coverage (Scenario 2)	L_{tree}	km	500
Shrub coverage (Scenario 2)	L_{shrub}	km	500
Crop coverage (Scenario 1 and 2)	L_{crop}	km	500
Tree root depth	$r_{d,tree}$	m	10
Shrub root depth	$r_{d,shrub}$	m	2
Crop root depth	$r_{d,crop}$	m	1
Length of wet season	τ	days	90
Atmospheric humidity at coast, wet season	q_{wet}	kg kg^{-1}	0.01
Atmospheric humidity at coast, dry season	q_{dry}	kg kg^{-1}	0.003
Loss of crop viability	β^*		0.25

Appendix 2: summary of potential critical transitions in the Amazon, consequences, indicators and monitoring options. From the Deliverable “A blueprint for an early warning for critical transitions system in Amazonia” of the AMAZALERT project.

class	Process	Causes (forcing type)[1]	Primary consequence (impact type)[2]	Ecosystem service affected	Indicator	monitor
Carbon cycle	Decrease GPP	Drought (3) High temperature (1) Nutrient loss (1)	NPP loss (2)	CO_2 sink Reduced productivity forest products Reduced appeal for tourism Cultural capital Air quality: Health (fire -> respiratory disease); Transport (smoke closing airports)	Degrading greenness, foliage, biomass	Flux data Regional [CO_2] Biomass plots Remote biomass Remote sensing multispectral
	Increase respiration	Drought (3) High temperature (1)	NPP loss (2) Soil carbon loss (1)		Degrading biomass Degrading soil carbon	
	Reduced recruitment	Gpp loss (1,3) Fire (3) Biodiversity change (1)	Biomass loss (2,3)		Opening canopies	Remote sensing secondary vegetation Permanent plots
	Increased mortality	Drought (3) Floods (3) Storms (3) Biodiversity change (1) Deforestation (2)	Biomass loss (2,3)		Increase of dead matter	Remote sensing of gaps and defoliation Permanent plots
	Increased fire incidence	Drought (3) Previous fire (1, 3) Mortality increase (1,3) Land-use change (2)	Biomass loss (3)		Degradation Opening canopies Fire frequency	Fire monitoring remote sensing Fire susceptibility: moisture, openness, fuel load
Water cycle	Reduced evapotranspirati on	Reduced rainfall Reduced GPP Reduced biomass	Reduced moisture transport and recycling Reduced/increased?/Change d? water stress	Regional water recycling Water provision for agriculture & other sectors, e.g. hydro power Flood protection Navigability -> access to markets and services Disease control	Degrading vegetation Increased river discharge	Flux data Catchment studies Surface temperatures
	Reduced precipitation	Global/regional climate effects Reduced evapotranspiration	Reduced soil moisture Reduced river discharge		Precipitation? (and related indicators, e.g. cumulative rainfall, drought indicators, refer to WMO Task Team on Climate Risk and	Rainfall network TRMM Soil moisture

[1] Numbering as in section 2.1 – first set
[2] Numbering as in section 2.1 – second set

class	Process	Causes (forcing type)[1]	Primary consequence (impact type)[2]	Ecosystem service affected	Indicator	monitor
					Sector-Specific Climate Indices) Lower river levels	
	Reduced runoff	Reduced evapotranspiration Reduced rainfall	Reduced river discharge		Lower river levels	River discharge
biodiversity	Changing competition	Drought High temperature Increased CO_2	Shift in species composition	Reduction in gene pools Reduced pollination service, affecting forest regeneration Effects on carbon sequestration? Tourism Cultural capital	Changes in key species abundance	Permanent plots Monitoring individuals
	Species loss	Habitat loss Over-exploitation Hunting	Degrading biodiversity		Changes in key species abundance	Inventories, plots, DNA fingerprinting
	Habitat loss	Deforestation Degradation Changing river levels	Degrading biodiversity		Changes in key species abundance	Remote sensing habitat inventory
Nutrients	fire	Deforestation Pasture burning	Slow recovery Changed vegetation type	Regrowth capacity of forests Agricultural fertility Air quality: Health (fire -> respiratory disease); Transport (smoke closing airports)	Degraded vegetation type	Soil inventories Remote sensing biomass density
	Topsoil erosion	Deforestation	Bare soils River siltation Slow recovery			
	Repeated logging	Economic demand	Soil degradation Slow recovery			
Economic	Financial capital	Total assets	Lack of financial resources for sustainability measures	All forest services	GDP growth	Production, investments
	Agricultural importance in Amazon economics	Areal extent (+)	Clear cut deforestation		GVA agriculture (%)	International trade
	Forest cover	Opening of forests (+)	Forest degradation		GVA Forestry	International trade
	Economic importance of Amazon	Exploitation of natural resources (+)	Natural resource depletion		Investment rates	International corporations
	Export	Production in Amazon (+)	Clear cut deforestation		Inflation rate (+/-) Production of ag. products (+)	International (meat) demand
	Forest cover	Value of standing forests (-)	Lack of forest protection		PES (0)	Ecosystem services valued
Social	Urban system	Urban population density (-)	Increase in inequality and poverty	Biodiversity, forest fragments	Rural outmigration (-)	Urban pull

class	Process	Causes (forcing type)[1]	Primary consequence (impact type)[2]	Ecosystem service affected	Indicator	monitor
	Rural system	Emptying of countryside (+)	Lack of social fabric in countryside		Labour force (-)	
	Legal system of protection	Illegal land ownership and deforestation (+)	Lack of control		Property rights (0)	Government control
		Illegal activities (+)	High rates of illegal activities		Control of corruption (0)	Government control
					Crime rates (0)	
	Education	Education of adults (0)	Behavioural change towards sustainable thinking		Illiteracy rate (0)	Government programs
		Education of youth (0)			School enrolment (0)	Access to school system
	Inequality	Income differences (+)	Rural poverty		Gini coefficient (0)	Multiple
	Income level	Low income (0)	Poverty		Poverty	Economy
	Health system	Health status (+)	Affects quality of life		Child mortality	Multiple
	Social capital	Quality of life (0)			Civil society involvement	