HOW LONG DO RESTORED ECOSYSTEMS PERSIST?[1]

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Abstract

Why do some restored ecosystems persist for centuries while others are quickly converted to alternative land uses or land covers? We propose that restored ecosystems have a temporal dimension that is variable, often finite, and likely predictable to some extent based on attributes of stakeholders, environment, and governance. The longevity of a restored ecosystem carries strong implications for its capacity to support biodiversity and provide ecosystem services, so an emerging challenge for restoration ecology is to predict the circumstances under which restored ecosystems persist for longer or shorter periods of time. We use a case study in tropical forest restoration to demonstrate one way that restored ecosystem longevity can be approached quantitatively, and we highlight opportunities for future research using restoration case study repositories, practitioner surveys, and historical aerial imagery. Much remains to be learned, but it is likely that decision-makers and practitioners have considerable leverage to increase the probability that restored ecosystems persist into the future, extending the benefits of contemporary restoration initiatives.

Key words: Ecological restoration, longevity, restoration success, survival analysis, tropical forest restoration.

When people designate land for restoration, ideally that land begins a recovery process that will continue in perpetuity without further degradation. In some cases, lands do recover for long time periods (e.g., Vallauri et al., 2002; Freitas et al., 2006), yet in many cases, lands recover to some degree and then are degraded again and repurposed for agriculture or other uses. For instance, many restored grasslands revert to crop fields when commodity prices are high (Secchi et al., 2009). The length of time that a site is allowed to recover carries strong implications for its capacity to provide habitat for biodiversity and benefits to society (Rey Benayas et al., 2009; Moreno-Mateos et al., 2012; Bayraktarov et al., 2016). Carbon storage, endangered species habitat, wild edible plants, and overall plant species richness are a few of the many attributes that tend to increase over time in regenerating ecosystems (Suganuma & Durigan, 2015; Crouzeilles et al., 2016; Sutherland et al., 2016).

Given the long time periods needed for most ecosystems to fully recover, an important question is: Why do some restored ecosystems achieve greater longevity than others? To the best of our knowledge, this question has not been addressed, although it is implicit in most conceptualizations of what constitutes restoration success (SER, 2004; Zedler, 2007; Le et al., 2012; McDonald et al., 2016). Here, we discuss some of the factors that could influence the expected longevity of a restored ecosystem, and we illustrate one quantitative approach to studying restored ecosystem longevity using a case study in tropical forest restoration.

Conceptual Framework

We conceive longevity to be the maximum age that a restored ecosystem attains before being converted to an alternative land use (Fig. 1). In the ideal situation when a restored ecosystem continues to persist as such, longevity is indefinite but bounded at the lower

[1] We thank the PARTNERS research coordination network, James Aronson, Robin Chazdon, Eva Garen, and David Moreno-Mateos for stimulating conversations and critical feedback on this concept. The case study experiment was funded by the National Science Foundation (DEB09-18112, DEB14-56520).

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doi: 10.3417/2017002

end by the time elapsed from the start of restoration to the present. In this context, a "restored ecosystem" is one that has been managed for ecosystem recovery, regardless of whether success has been achieved (Zedler, 2007). The concept of longevity can be applied to many kinds of ecosystems, and many land uses falling under the banner of restoration, sensu lato (McDonald et al., 2016; Aronson et al., 2017). Here, we focus on restored forests, reflecting our collective expertise as well as the international movement to restore forests and forested landscapes globally (UNCBD, 2012; UNCCD, 2015; UNFCCC, 2015; IUCN, 2016).

We propose that the main factors that influence restoration longevity are stakeholder preferences and capabilities, environmental attributes, and the rules of governance that influence the relationship between stakeholders and the environment. Stakeholders are people who have a vested interest in a restored ecosystem and some control over how it is managed; these may often be landowners, but can also include organizations, institutions, and communities. Specific predictions about the influence of stakeholders, environment, and governance are provided in Table 1; each of these factors is subject to temporal change, interactions, and feedback from restoration activities.

One general prediction is that restored ecosystems are likely to persist for longer periods of time when stakeholders have long-term restoration goals and sufficient resources to pursue them, including technical capacity and funding (Holl & Howarth, 2000; Le et al., 2014). For instance, forest rehabilitation projects in the Philippines received more upkeep (fire breaks and forest patrols) and were re-cleared less often when they had long-term maintenance and monitoring plans (Chokkalingam et al., 2006). Yet, long-term monitoring plans are frequently neglected (Holl & Cairns, 2002; Murcia et al., 2015). By the same token, forests restored with short-term financing plans are likely to face strong economic incentives for conversion after funding resources or incentives have expired (Lamb, 2014). Many forest restorations financed by carbon markets are committed to maintaining a "permanent" carbon reservoir for only 20 to 50 years, for example (Galatowitsch, 2009)—decades less than the time required to saturate carbon sequestration in many regenerating forests (Chazdon et al., 2016).

In addition, environmental factors may interact with stakeholder decisions, or in some cases constrain restoration longevity independently. The speed of forest recovery, for instance, can affect how stakeholders perceive their options with regard to regenerating forest. In Costa Rica, grassy restoration areas with low tree cover were sometimes viewed as "unused," thereby putting them at risk of being co-opted for cattle grazing during the dry season, effectively resetting secondary succession (Zahawi et al., 2014). Large-scale natural disturbances can also force land use or land cover transitions in restored ecosystems, either directly (e.g., climate-related forest diebacks accelerated by massive wildfires in the western United States; Falk, 2017) or indirectly (e.g., when climate change makes previously unsuitable areas fit for profitable agriculture; Titeux et al., 2016). Variance in the susceptibility of restored ecosystems to such disturbances will manifest as variance in their longevities.

Stakeholder-environment interactions, including restoration, are mediated by governance, which can influence the prospects for restored ecosystem longevity indirectly through rules, incentives, and restrictions. Perhaps the clearest influence of governance is through land tenure systems. When government policies cause stakeholders to lack confidence in their rights to use or transfer land (e.g., due to lack of legal title or a history of land grabbing; Byron, 2001), stakeholders are generally unwilling to begin or continue investing in any long-term land use, including restoration and forest conservation (Unruh, 2008; Lamb, 2014; Mansourian & Vallauri, 2014; Robinson et al., 2014).

Challenges to restored ecosystem longevity are likely to be additive, interactive, and temporally dynamic. Deforestation moratoria, for example, are government regulations that limit forest clearing based on variables such as canopy cover and forest height, which vary by forest age and forest type (e.g., Costa Rica, 1996). When a restoration project produces forest that meets the legislated criteria, reverting to another land use becomes legally complicated. In northeastern Costa Rica, Fagan et al. (2013) estimated that some forests may reach these cutoffs after eight to 12 years of recovery, and they showed empirically that older native reforestations were cleared at lower rates than young reforestations following the passage of the law (Fig. 2). However, the law may have also created a perverse incentive to clear younger second growth before it matures (Sierra & Russman, 2006), highlighting the fact that policies not only interact with ecosystem resilience but can also influence restoration longevity differentially over different timescales.

Case Study: Longevity of Restored Tropical Forest in Costa Rica

To illustrate how longevity can be approached quantitatively, we draw on a tropical forest restoration experiment in southern Costa Rica. Between 2004

Table 1. Attributes contributing to restored ecosystem longevity and the time frames over which they are expected to exert influence.

Attributes	1–10 yrs.	10–100 yrs.	100–1000 yrs.	References
Governance attributes				
Land designations (e.g., protected area, indigenous territory)	+	+	+	Andam et al. (2008); Nolte et al. (2013); Carranza et al. (2014); but see Mascia & Pailler (2011)
Land tenure system (e.g., significance and history of tenure status)	+	+	+	Byron (2001); Le et al. (2012)
Legal restrictions and incentives (e.g., deforestation moratoria, payments for ecosystem services)	+	+	+	Sierra & Russman (2006); Fagan et al. (2013)
Stakeholder attributes				
Community engagement	+	+		Bass et al. (1995); Higgs (2003); Pulhin & Pulhin (2003); Le et al. (2012); Mansourian & Vallauri (2014); Wilson (2015); Lazos-Chavero et al. (2016)
Effective leadership	+	+		Gooch & Warburton (2009); Le et al. (2012)
Technical capacity (e.g., species-site matching, site preparation)	+	+		Stanturf et al. (2001); Le et al. (2012)
Land tenure security	+	+	+	Byron (2001); Oviedo (2005); Unruh (2008); Lamb (2014); Mansourian & Vallauri (2014); Robinson et al. (2014)
Organizational resilience	+	+	+	Gooch & Warburton (2009)
Resources (e.g., current and projected funding)	+	+	+	Holl & Howarth (2000); Galatowitsch (2009); Brancalion et al. (2012); Martin (2016)
Long-term vision (e.g., monitoring and adaptive management plans)		+	+	Holl & Cairns (2002); Vallauri et al. (2002); Chokkalingam et al. (2006); Le et al. (2012); Mansourian & Vallauri (2014)
Environmental attributes				
Ecosystem resilience (e.g., speed of recovery)	+	+		Holl & Aide (2011); Fagan et al. (2013); Zahawi et al. (2014)
Intensity of past land uses	+	+		Stanturf et al. (2001); Chazdon (2008); Holl & Aide (2011)
Climate change susceptibility (e.g., projected forest dieback)	+	+	+	Falk (2017); Williams et al. (2007); Allen et al. (2010)
Disturbance regime (e.g., stand-replacing floods, fires)	+	+	+	Stanturf et al. (2001); Falk (2017)
Landscape context (e.g., proximity to seed sources, roads)	+	+	+	Stanturf et al. (2001); Laurance et al. (2002); Jacquemyn et al. (2003); Dunwiddie et al. (2009); Wyman & Stein (2010); Holl & Aide (2011); Tambosi et al. (2013); Crouzeilles & Curran (2016)
Suitability for alternative land uses (e.g., mining, agriculture)	+	+	+	Latawiec et al. (2015); Titeux et al. (2016)

and 2006, 54 experimental restoration plots (0.25 ha each) were established at 18 sites on former cattle pastures in the premontane wet forest zone between the Las Cruces Biological Station (8°47′N, 82°57′W) and the town of Agua Buena (8°44′N, 82°56′W) in Coto Brus County (1100–1400 m.s.m., ~3–4 m precip. yr.$^{-1}$). Each site contained three plots, which were randomly assigned one of three restoration treatments: natural regeneration, applied nucleation (small patches of trees planted to mimic natural succession), and tree plantations (for details see Holl et al., 2011). This experiment was done in collaboration with private landowners; 36 plots (12 sites) were established on leased private lands owned by Costa Rican farmers, and 18 plots (six sites) on lands owned by North Americans (who received no financial compensation) or by the Organization for Tropical Studies, a nongovernmental organization. During the course of the experiment, 12 plots (four sites) changed ownership, from Costa Rican to North American; three of these would have been converted to agriculture had they not been purchased. Rental agreements for restoration plots on leased farmlands were made for 5-year periods, and landowners were

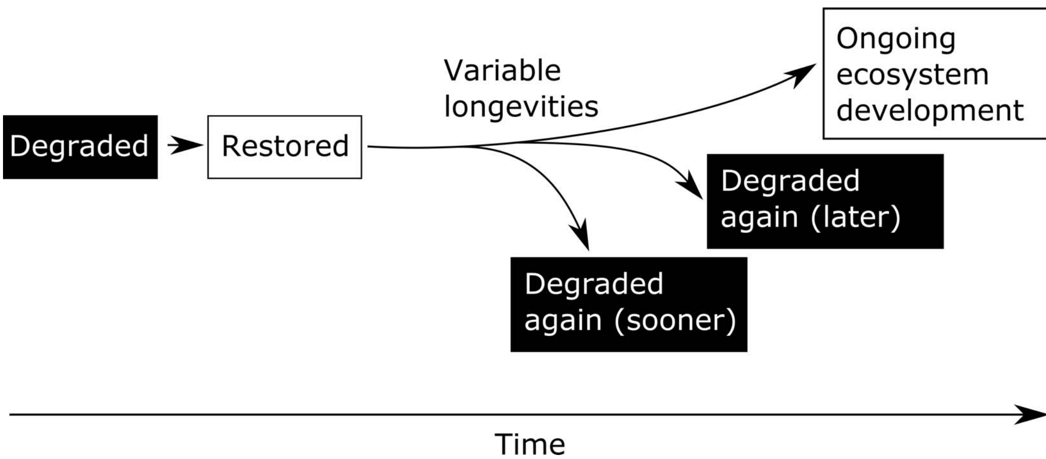

Figure 1. Restored ecosystem longevity. To operationalize this concept for forests, "degraded" can be replaced with "non-forest" and "restored" can be replaced with "forest."

paid approximately what they might have made farming cattle (\$150–400 USD ha^{-1} $yr.^{-1}$).

Analyzing restoration longevity was not the original purpose of this experiment; the purpose was to test a novel strategy for tropical forest restoration. However, over the decade that this project has been running, some plots were deforested and/or converted to alternative land uses (e.g., cattle pasture, banana plantation), providing an opportunity to evaluate longevity. Longevity calculations were made as of summer 2016; thus, maximum potential longevity for any plot is 10 to 12 years. We analyzed longevity using Kaplan-Meier survival analysis with log rank tests (Appendix 1).

In the 10 to 12 years since this experiment began, 36 restoration plots (67%) continued to recover, but 18 plots (33%) were deforested and/or converted back to agricultural land uses. Average longevity was 9.9 ± 4.1 years (mean ± SD). Plot conversions were evenly distributed among restoration strategies (six natural regeneration, seven applied nucleation, five plantations), but Costa Rican farmers under rental contracts converted land at higher rates than other landowners, resulting in significantly shorter periods under management for forest recovery (Fig. 3). The difference in restoration longevity between lands owned by North Americans (11.8 ± 1.3 years) and Costa Ricans (7.4 ± 5.1 years) probably reflects differing views on the economic value of agricultural land; many Costa Rican landowners received some or most of their income from farming, whereas foreigners did not. In addition, there may have been an interaction between ownership and the potential suitability of the land for agriculture, as lands owned by North Americans tended to be more severely degraded (pers. obs.). Moreover, this case study highlights the importance of local buy-in for restored ecosystems to persist (Murcia et al., 2015).

Discussion

Our premise is that restored ecosystems have a temporal dimension that is variable, often finite, and likely predictable to some extent based on attributes of stakeholders, environment, and governance. Whereas longevity is important to the total value of

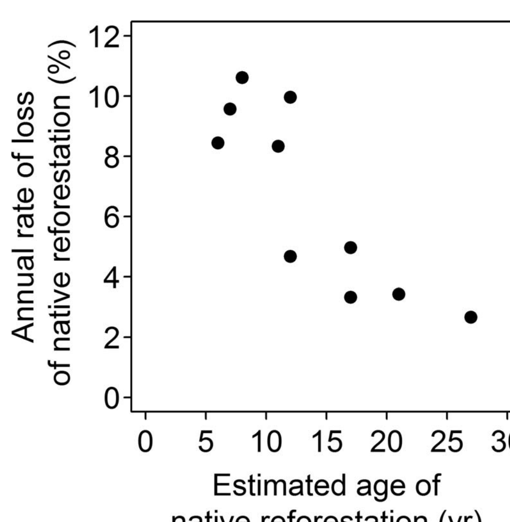

Figure 2. Relationship between native reforestation age in years and the annual rate of loss of native reforestations in northeastern Costa Rica, a country with a deforestation ban. Native reforestation includes natural forest regeneration and tree plantations using native tree species. Source: Fagan et al. (2013).

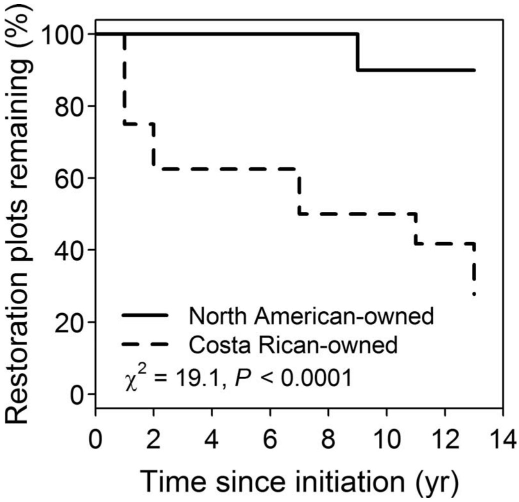

Figure 3. Effect of land ownership by Costa Rican farmers ($N = 24$ plots) versus North Americans ($N = 30$ plots) on the longevity of restored tropical forests in southern Costa Rica. Four sites that transferred ownership from Costa Rica to North America during the experiment are coded here as North American, reflecting their final ownership. See Appendix 1 for details.

a restored ecosystem over its lifetime, there will certainly be cases where the need for restoration is great and immediate but the potential for longevity is limited. For instance, planting a Great Green Wall across the northern Sahel entails serious technical and societal challenges in a demanding environment (Sacande & Berrahmouni, 2016), but the alternative—desertification of large parts of Africa—has severe social and environmental ramifications. In such cases, it is worth considering that there are probably multiple pathways to achieving long-term restoration; restoration projects with low longevity potential by one measure may still have improved prospects by other means. In the highlands of Madagascar, for example, land tenure and funding continuity are precarious, and grassland fires are an annual threat to regenerating forests; but these limitations may be overcome by strong community support, as when 200 villagers self-organized to put out a forest fire at the Ankafobe restoration area (C. Birkinshaw, pers. comm.).

A potential criticism of this line of research is that disturbance and ecological succession are cyclical, and therefore, restored ecosystem longevity should not be idealized in a way that ignores or excludes natural disturbance regimes. We note that restored ecosystem longevity is not necessarily equivalent to "time since disturbance." Restoration is a human-environment relationship that can and often does span significant disturbances, some of which are critical to ecosystem development (e.g., fire in Missouri, U.S.A., woodlands; McCarty, 1998). Additionally, long-undisturbed ecosystems sometimes possess rare and unique values (e.g., habitat for rare species; Dunk & Hawley, 2009), and in such cases it may be important to maintain not only the temporal continuity of the land use (i.e., restoration) but also of the land cover (i.e., undisturbed forest), particularly when disturbances are large and severe while restored ecosystems are small and at risk of population extirpations.

If restored ecosystem longevity is to be pursued, an emerging challenge for restoration ecologists is to develop predictive models, as medical researchers have done to improve outcomes in human longevity (Passarino et al., 2016). We used a simple, illustrative example from a well-documented, replicated experiment, but future work will require more diverse and more representative cases. One source for these data may be historical aerial imagery; sequences of images can reveal when some ecosystems (e.g., forests) emerged, persisted, and were cleared over large areas (e.g., Zahawi et al., 2015). Digital repositories also house large collections of restoration case studies (ELTI, 2016; SERI, 2016), which could serve as a starting point for longevity studies. There are many examples of restoration projects started less than 100 years ago, and centenarian projects are less abundant but cases do exist (Vallauri et al., 2002; Freitas et al., 2006). Finally, many restoration practitioners will know of restored sites that have persisted or were converted over varying time periods, but documenting these events could be challenging since researchers and practitioners alike prefer not to highlight unsuccessful projects (Zedler, 2007; Suding, 2011). A key consideration for future studies will be identifying a statistical sample that is unbiased by the tendency for longer-running and more successful projects to be more detectable (Lortie et al., 2007).

Much remains to be learned, but it is likely that decision-makers and practitioners have considerable leverage to increase the probability that restored ecosystems persist into the future. Locally, practitioners can engage communities to build stakeholder support and facilitate training to improve technical capacity. Programmatically, project managers can prioritize restoration in sites to minimize competition for alternative land uses (Latawiec et al., 2015). And at a national scale, politicians can pass legislation that incentivizes long-term management and penalizes destructive activities. Moreover, international restoration commitments are currently dominated by hectare-based pledges to restore large areas of young forest by 2020 or 2030 (IUCN, 2016), but if restored ecosystem longevity is at least partially controllable, then an ambitious, confident country could go even farther. A truly long-term commitment would be to restore a million hectares of 100-year-old forest by 2120, or a million hectares of 300-year-old forest by 2320.

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Appendix 1. Survival analysis.

We used Kaplan-Meier survival analysis with log rank tests (Harrington & Fleming, 1982) to evaluate differences in forest restoration longevity between restoration strategies and under different ownerships (Costa Rican vs. North American). If discontinuations were randomly assigned, 12 would be expected on lands owned by North Americans and six on lands owned by Costa Ricans, but 15 (83%) of discontinuations occurred on leased lands owned by Costa Ricans ($\chi^2 = 19.1$, $P < 0.0001$). To account for pseudoreplication from multiple plots within sites with a single landowner, we repeated the analysis for applied nucleation plots only, with the same result ($\chi^2 = 8.7$, $P = 0.0032$). During the course of the experiment, 12 plots (four sites) changed ownership, from Costa Rican to North American; three of these would have been converted to agriculture had they not been purchased. These four sites are coded as "North American" here, reflecting their final ownership. Survival analysis was implemented in the survival package (Therneau, 2015) in R version 3.3.0 (R Core Team, 2016).