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REVIEW ARTICLE

Undertaking large-scale forest restoration to generate ecosystem services

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The global community is seeking to substantially restore the world's forest cover to improve the supply of ecosystem services. However, it is not clear what type of reforestation must be used and there is a risk that the techniques used in industrial timber plantations will become the default methodology. This is unlikely to be sufficient because of the well-known relationship between biodiversity and ecological functioning. Restoration may be achieved through natural regeneration but this may not always occur at critical locations. Ecological restoration involving species-rich plantings might also be used but can be difficult to implement at landscape scales. I review here the consequence of planting more limited numbers of species and the effects of this on the delivery of ecosystem services. Evidence suggests many commonly sought ecosystem services—though not all—may be generated by the modest levels of species richness provided these species have appropriate traits. The literature also shows that the alpha diversity of restored forests is not the only driver of functionality and that the location and extent of any reforestation are significant as well; beta and gamma diversity may also affect functionality but these relationships remain unclear. Encouraging the adoption of even moderately diverse plantings at landscape scales and at key locations will require policies and institutions to balance the type, location, and scale of restoration and make the necessary trade-offs between national and local aspirations. New approaches and metrics will have to be developed to monitor and assess restoration success at these larger scales.

Key words: biodiversity, ecosystem functioning, ecosystem services, Forest landscape restoration, rehabilitation, scale

Implications for Practice

- It is not necessary to restore all the original biodiversity in order to generate many ecosystem services. This should make it easier to undertake functionally effective restoration on a large scale.
- Many ecosystem services are also dependent on the particular traits represented in the species assemblage. Identifying these is a critical future task.
- The extent to which ecosystem services are generated depends, as well, on the spatial extent and location of any restoration. This means that new assessment tools will be needed to monitor the collective effectiveness of different combinations of forest type, extent, and location.
- New policies and institutions will be needed to make restoration attractive and facilitate the many trade-offs necessary to achieve satisfactory outcomes for all stakeholders.

Introduction

There is increasing global interest in restoring forests on a large, landscape scale. The New York Declaration on Forests has set a target of seeking agreement by 2030 to restore 350 million hectares (Bonn Challenge 2014), the U.N. Convention on Biodiversity aims to restore 15% of all degraded ecosystems by 2020 (CBD 2011), and the U.N. Sustainable Development Goals (Goal 15) seeks to restore degraded land and achieve a "land-degradation neutral world" by 2030 (SDG 2015). In recent years, a growing number of large-scale regional and national reforestation goals have also been established (Lamb 2014). These programs pose a challenge; most reforestation over the last 100 years has been carried out to produce timber but the primary motive for these recent reforestation goals is not to just produce timber but to conserve biodiversity and increase the supply of ecosystem services once provided by natural forests. That is, a paradigm shift is underway and new forms of reforestation will be needed to achieve these new global aspirations. In what follows, I will use the term restoration to describe all forms of multispecies reforestation on deforested areas while reforestation remains a more general term covering monocultures as well as multispecies cultivation.

Two countries that have recently undertaken reforestation on very large scales are China and Vietnam. An objective

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in both cases was to improve the supply of ecosystem services, especially those associated with watershed protection (Chokkalingam et al. 2006; De Jong et al. 2006). However, both countries have relied heavily on reforestation methods developed to establish industrial timber plantations simply because the species and methodologies are well-known. Timber plantations involving a single species can certainly provide some ecosystem services depending on the species used, the intensity of management, and the rotation length (Brockerhoff et al. 2008; Lamb 2014). But their capacity to generate many of the ecosystem services now being sought is often likely to remain modest. Monocultures will continue to play a role in future global reforestation programs but other options are also needed.

One widely discussed alternative to reforesting large deforested areas is to facilitate natural regeneration (Chazdon & Uriarte 2016; Holl 2017). Where it occurs, natural regeneration is attractive because it reduces costs and solves the problems of seed sourcing and species-site matching (Nunes et al. 2017). Forests resulting from natural regeneration can restore considerable functionality and many ecosystem services. But natural regeneration does not always occur, and the diversity present may be limited, depending on the site history and landscape context. This can be problematic at key locations within landscapes that are critical for the improvement of ecosystem functioning. In addition, natural regeneration may be difficult to protect long enough for it to be effective since many in the community may see these sites not as regrowth forests but as wastelands and therefore available for other purposes (Lamb 2014).

Many ecologists point to ecological restoration as being an ideal solution (McDonald et al. 2016). Those adopting ecological restoration usually aim to restore most of the species formerly present and this is seen as advantageous because of evidence of a positive relationship between species richness, ecosystem functionality, and the supply of ecosystem services (Harrison et al. 2014; Barral et al. 2015; Winfree et al. 2015). But the majority of these studies have been carried out over much smaller areas than are now being discussed by the global community (de Souza Leite et al. 2013; Holl 2017). Scaling-up could be difficult because most landscapes contain many separate ecosystems, each with its own assemblage of species. Little may be known about many of these target ecosystems or about the environmental requirements of their constituent species. Furthermore, few may still be represented by residual areas able to act as reference sites and seed of the many species involved may be hard to locate. In short, ecological restoration at these large scales is likely to be difficult and expensive.

A third option might be to plant some, but not necessarily all, of the original tree species. This would certainly simplify the task (and cost) of reforestation. But would it be functionally effective? And if it is to be used, then which parts of a landscape are most critical and how much reforestation must be carried out at these sites? Note that while multifunctionality is desirable, many landholders, as well as funding bodies, are more interested in some ecosystem services than in others.

I report here on (1) evidence concerning the relationship between the number of tree species used in reforestation and the generation of ecosystem services, (2) how biodiversity conservation and the supply of ecosystem services might be affected by where in a landscape these various forms of reforestation are carried out and the scale at which it is done, and (3) how reforestation at these large landscape scales might be carried out in ways that are attractive to landholders.

Tree Species Richness and the Generation of Ecosystem Services

Three hypothetical relationships between the number of tree species planted and the generation of specific ecosystem services are shown in Figure 1. Line A indicates that some ecosystem services are largely restored when only limited numbers of the former species are restored while lines B and C show that moderate or higher levels of richness are required before particular ecosystem services are generated.

Evidence concerning the nature of the relationships for various ecosystem services is outlined in Table 1. Cases where ecosystem services are generated by monocultures or by restoring relatively limited numbers of species (line A) include carbon sequestration, soil fertility improvement, soil erosion control, and the restoration of habitats for wildlife that are habitat generalists. By contrast, cases where ecosystem services require a much higher number of species (line C) include the provision of habitats for wildlife that are habitat specialists, or the restoration of habitats to sustain species able to provide pollination services and pest control in adjacent agricultural areas. The restoration of

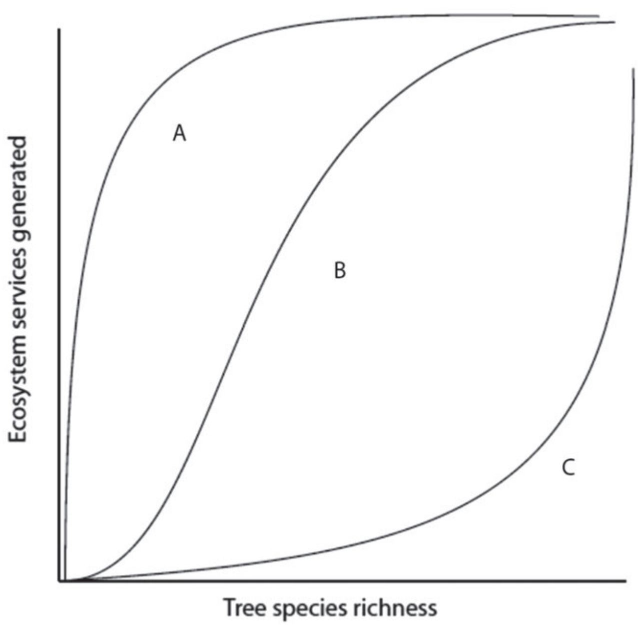

Figure 1. Three hypothetical relationships between the numbers of tree species planted at a particular site and the generation of ecosystem services. Line A indicates that some ecosystem services are largely restored when only limited numbers of the former species are restored while lines B and C show that moderate or higher levels of richness are required before particular ecosystem services are generated.

Table 1. The relationship between increasing tree species richness, the traits of these species, and the provisioning of ecosystem services.

Ecosystem Services Provided by Forests	*Response to Increasing Species Richness (see Fig. 1)*	*Types of Species and Functionally Important Traits Needed*
Soil protection–water erosion	A–B	Favored by species producing thick litter layers and having deep, widespread, and fibrous roots able to stabilize soils. This combination not always found in a single species. But also enhanced by using species with open canopies or planted at lower densities, which allow development of a ground cover (Pohl et al. 2009; Genet et al. 2010; Marden 2012; Wang et al. 2014).
Soil protection–wind erosion	A	Any species tolerant of exposed sites will be beneficial and may also facilitate subsequent successional development (Yoshikawa 2010; Miyasaka et al. 2014). Trees used in windbreaks should have deep but porous crowns and, preferably, short laterally spreading roots that limit competition with agricultural crops or pastures (Tamang et al. 2010).
Restoring soil fertility	A–B	Soil nitrogen improved by nitrogen fixing species; other soil properties can be improved by deep-rooted species able to grow well in compacted or infertile soil (Wang et al. 2010; Singh et al. 2012; Gei & Powers 2013; Wang et al. 2013).
Annual water yield	A–B	Where mixtures are more productive than monocultures they are likely to use more water and therefore reduce annual water yields; but water use also depends on the traits of the constituent species as well as tree ages and tree densities rather than on richness alone (Kunert et al. 2012; Schwendenmann et al. 2015; Lübbe et al. 2016).
Improved dry season stream flow	(A–B)?	Species may differ in their capacity to improve infiltration rates in degraded soils but there is only circumstantial evidence that mixtures are more effective than monocultures (Ilstedt et al. 2007; Bonell et al. 2010; Lacombe et al. 2015).
Reduced incidence and severity of flooding	Nil	No evidence that species richness affects flooding.
Salinity control	A	Any fast-growing trees having high rates of evapotranspiration can help lower water tables (George et al. 2012).
Habitats for species conservation	A, B, or C	Monocultures may suit some habitat generalists but wildlife with more specialized habitat requirements will require more structurally complex forests with a diversity of foliage, flowering, and fruiting resources (Cork et al. 2000; Kavanagh et al. 2007; Ren et al. 2007; Lugo et al. 2012). There is only weak evidence that increased tree species diversity improves soil fauna diversity (Korboulewsky et al. 2016).
Control of pests of adjacent agricultural crops	B–C	Control most likely in structurally complex forests able to provide year-round habitats for a greater variety of predators and parasitoids of agricultural pests; the habitat requirements will vary with the specific pest species. There is the possibility that these diverse forests may also provide habitats for crop pests (Bianchi et al. 2006; Railsback & Johnson 2014).
Pollination of agricultural crops	B–C	Sufficient tree species are needed to attract and sustain pollinators by providing habitats and ample pollen and/or nectar throughout the year (Menz et al. 2011; Jha & Kremen 2013; Ponisio et al. 2015). Note that some pollinators may be more critical than others (Winfree et al. 2015)
Carbon sequestration	A–C	Monocultures of fast-growing species can achieve rapid carbon uptake but long-term sequestration is likely to be favored by multispecies plantations including those with tall trees having high wood densities; such mixtures are also likely to have the resilience that long-term carbon storage requires (Conti & Díaz 2013; Hulvey et al. 2013).
Aesthetics and recreation	B–C	Favored by having a variety of life forms, foliage shapes, and colors (Dallimer et al. 2012; Carrus et al. 2015).

a higher level of species richness is also an advantage when the objective is to improve landscape aesthetics or provide recreational opportunities. Between these two patterns are a large number of ecosystem services that can be supplied by restoring some, but not necessarily all, of the former species richness. In most cases, the supply of these services increases in proportion to any increase in species richness (indicated by line B in Fig. 1). Examples of these relationships are soil conservation, watershed protection, and the restoration of soil fertility. Not all of the relationships are clear and many are also influenced by species traits, the relative proportions of species used in planting mixtures as well as by planting densities (which affect the extent of understory development). The landscape context is also important (see further below).

An example of the uncertainties in these relationships is provided by the relationship between diversity and hydrological processes. Annual water yields and dry season flows are often of major concern to those undertaking restoration. But the influence of species richness on hydrological processes is complicated; any kind of restoration will increase evapotranspiration and diminish stream flows but, at the same time, restoration of severely degraded lands may also improve water infiltration rates, increase groundwater stores, and reduce overland flows. The impact on baseflow and whether or not dry season streamflow improves then depends on the relative importance of these two processes (Bruijnzeel 2004). Some evidence suggests that multispecies plantings may use more water than monocultures and thus reduce both annual and seasonal water flows (Table 1). But other studies have not found this relationship. Likewise, the effect of multispecies plantings on the infiltration capacity of soils also remains unclear (Table 1). However, the net impact depends as well on the proportions of different species used and their transpiration rates, differences in growth rates and rates of rainfall interception by canopies and litter, as well as differences in understory biomass and the way forests are managed (Lacombe et al. 2015). Finally, hydrological relationships are also influenced by location and scale (see further below). In summary, it is possible that planting a simple species mixture may improve dry season flow in heavily degraded sites but, more generally, it seems that hydrological flows are not as strongly influenced by species richness as by a variety of other factors. The issue obviously deserves further investigation.

These same uncertainties also exist in our understanding of the processes underlying the delivery of many other ecosystem services as well (Birkhofer et al. 2015). Reflecting this, the relationships in Table 1 are shown as A–B, B–C, or A–B–C depending on the evidence that is currently available. This means that, of the 12 ecosystem services reviewed, there was evidence for relationship A in 8 cases, for relationship B in 9 cases, and relationship C in 5 cases. In short, relatively modest numbers of tree species are likely to generate a number of ecosystem services although different species may be more effective than others depending on their traits and on the service required. But the evidence also suggests a wider variety of ecosystem services will be generated when a larger number of species are used. Interestingly, van der Plas et al. (2016) have noted that more species can generate a greater degree of multifunctionality when only moderate levels of functioning are required but that this diversity may be negatively related when higher levels of functioning are required for certain services. Again, much depends on the species traits.

Spatial Location of Restoration and the Generation of Ecosystem Services

Despite the prominence given in the literature to biodiversity–ecological functioning relationships, the diversity of tree species is not the only factor influencing the extent to which restoration generates ecosystem services. Locations are also important because landscapes are not uniform and some locations are more critical than others for certain ecosystem services (Table 2). Note that some of these critical sites may occupy a large proportion of a particular landscape (e.g. steep lands) while others, though functionally important, may occupy only small areas.

The spatial location of restoration is obviously important for reducing erosion but it is also influences hydrological flows. For example, in subhumid areas, tree planting on foot slopes or in riverine locations has a much larger effect on groundwater use than when carried out at locations further away from the streams due to differences in the proximity of the water table (Vertessy et al. 2003). These locational differences have been used to combat dryland salinity by accelerating evapotranspiration to lower water tables and one study found the careful location of reforestation could increase its effectiveness by up to seven times over more random placements (van Dijk et al. 2007).

Location also matters when restoring species habitats. Restoration around existing natural forest remnants can effectively enlarge these and help protect the biota these contain (Jones et al. 2016). In such cases, priority might be given to locations providing habitats of threatened species (Tobón et al. 2017). Biodiversity conservation is also assisted by new forests forming habitat corridors, which link patches of remnant natural forest and allow biota to move across landscapes or provide altitudinal or latitudinal pathways (Worboys et al. 2010). These also improve overall resilience by enabling forest communities to adapt to changing environmental conditions. The location of these corridors is critical, but so too is the type of connectivity (including the width and the type of new forest) as well as the scale over which they operate (Mitchell et al. 2013).

It is not easy to determine the priority that should be given to these different locations. Some will be important for a particular ecosystem service while others may generate multiple ecosystem services. In addition, the benefit-to-cost ratio of restoration is also likely to vary. This means different stakeholders are likely to have different priorities. The ways the trade-offs might be made are discussed further below.

Scale of Restoration Needed to Generate Ecosystem Services

The third factor influencing the supply of ecosystem services is the scale at which restoration is carried out, both overall and

Table 2. Locations within landscapes where restoration can help generate various ecosystem services.

Desired Ecosystem Service	*Key Location*	*Mechanism*
Control of erosion	Steep slopes	By stabilizing soil surfaces and impeding soil movement (Vanacker et al. 2007; Wang et al. 2012).
Control of sedimentation, improved water quality	Riverine areas	By reducing stream bank erosion and limiting soil and nutrient movement into water bodies (Feld et al. 2011; Asbjornsen et al. 2014).
Improved dry season water flows	Sites with compacted soils	By improving the infiltration capacity of surface soils (Bruijnzeel 2004; Ilstedt et al. 2007).
Control of dryland salinity	Semiarid agricultural landscapes	By increasing evapotranspiration and lowering water tables (Stirzaker et al. 2002; van Dijk et al. 2007).
Maintenance of water yield while improving other forest ecosystem services	Upslope areas away from riverine areas in subhumid landscapes	Planting away from streams limits root access to water accumulating in lower landscape positions and reduces impact on stream flow (Vertessy et al. 2003). Limiting reforestation to <20% watersheds limits impact on water yields (Bruijnzeel 2004).
Wildlife habitats	Around residual forest patches	By providing a protective buffer zone, reducing edge-effects and enlarging habitat areas to make them more attractive to resident species as well as more mobile species; likely to be especially useful around patches representing potential climate change refugia (Thomson et al. 2009; Jones et al. 2016).
Wildlife habitats	Between residual forest patches (i.e. corridors)	By providing connectivity between habitats and meta populations (Chetkiewicz et al. 2006).
Pollination, pest control	Within or near agricultural croplands	By providing habitats for pollinators and predators of pests (Tscharntke et al. 2005; Bianchi et al. 2006; Mitchell et al. 2013).
Facilitating climate change amelioration via atmospheric carbon storage	Former forested areas with low land prices and with soils and climate that are conducive to tree growth	Locations likely to be sensitive to carbon prices and opportunity costs of restoration (Polglase et al. 2013; Carwardine et al. 2015).
Facilitating climate change adaption	Altitudinal and latitudinal corridors between remnants; buffers around putative refugia	By facilitating species movement; these can create buffer zones and enlarge putative refugia using tree species and genotypes likely to be tolerant of future conditions (Vos et al. 2008; Sgro et al. 2011; Jones et al. 2016).
Protection against storms and tsunamis	Coastal areas	By forming protective forest buffer zones (Stanturf et al. 2014).
Multiple	Abandoned former agricultural lands	By increasing forest cover over large areas by planting or facilitating natural regrowth.

at particular locations. Much depends, of course, on the extent of past deforestation. Scale is important because many ecological processes largely operate at landscape scales (Holl et al. 2003). In most situations, it is unlikely that forest cover can be completely restored because of the need to maintain other land uses such as agriculture. An important question, therefore, is whether there is some threshold forest cover that must be exceeded before a particular ecosystem service is generated? It seems that thresholds often exist but they are generally poorly defined and, in any case, are likely to differ for different ecosystem services (Johnson 2013). In the case of thresholds for species habitats, estimates have ranged widely depending on the species being studied (Johnson 2013; van der Hoek et al. 2015). One meta-analysis suggested restoration success (i.e. recovery of species numbers, abundance, and distribution) is greater and more certain when contiguous forest cover exceeds 50% (Crouzeilles & Curran 2016). This may be an impossible target in many extensively deforested landscapes but it is clear that more forest is likely to enable the conservation of a wider range of species, especially if the structural complexity of the new forests resembles that of the former forests. More specific threshold areas will apply when habitats are being restored to support particular species.

The relationship between areas restored and hydrological flows is equally complex. Studies in small watersheds suggest that reforestation must exceed 20% of the area before any negative impact on water yields can be detected (van Dijk & Keenan 2007). Beyond this threshold, further increases in forest cover generally lead to declines in water yield. But the negative impact of reforestation on water yields at larger spatial scales is less clear. Some studies in large watersheds find water yields decline following large-scale reforestation (Li et al. 2014). Especially large negative impacts of reforestation on streamflow have been reported for subhumid and semiarid areas, such as northern China where the climate does not support the occurrence of natural forests (Feng et al. 2012). However, other studies have found the impact declined at larger scales (Filoso et al. 2017) or report no such changes, at least in dry season flows (Beck et al. 2013). Most of these studies separated the year-to-year variation on streamflow from the effects of land cover changes using hydrological modeling approaches but the reliability of such analyses is severely constrained by uncertainties in areal rainfall inputs. As noted earlier, reforestation can also have positive or negative effects on water yield depending on whether it improves infiltration rates sufficient to compensate for increased evapotranspiration. This balance may differ in different parts of the landscape.

Forests can be restored across a landscape in a number of smaller patches at critical locations or in a few large areas that cover the same overall area. Does this difference affect the generation of particular ecosystem services or the variety of such services? As noted earlier, there is no doubt than some wildlife species require large and contiguous habitat areas. Likewise, extensive restoration may be needed in hilly landscapes to control erosion. But interventions to restore small areas may have some advantages (Asbjornsen et al. 2014). One is that multiple patches scattered across an agricultural area can be beneficial for biota that perform critical functions such as pollination or pest control since these will then have shorter dispersal distances into adjacent croplands (Kremen & M'Gonigle 2015). Likewise, a landscape with many separate new restoration areas may be able to represent and sustain a more complete sample of the ecological niches and communities in that landscape than a single new forest block, especially if these patches are not too isolated.

But large scale of restoration beyond certain thresholds can sometimes generate negative as well as positive outcomes. For example, restoration of farmland in long-established cultural landscapes may lead to the loss of uncommon species of conservation significance now adapted to these habitats and their replacement by relatively common generalist species adapted to the new forest habitats (Navarro & Pereira 2012).

Discussion

Implications for Public and Private Landholders

Governments may be keen to promote forest restoration but any large-scale program is likely to involve land held by private landholders as well as public lands. These private landholders are likely to have different objectives and priorities than governments and their interest in reforestation as a way of generating ecosystem services (rather than, say, timber) will depend on what they perceive its opportunity costs to be. This means, firstly, that a variety of reforestation methods are likely to be used and, secondly, that some pragmatism will be needed when making choices.

Two examples illustrate the way such pragmatism can operate. The first concerns a large-scale forest restoration program being implemented in Hunan Province in China by the provincial government (World Bank 2012). The project covers a collective area of 60,000 ha and the primary objective of restoration is to establish resilient ecosystems able to protect watersheds, reduce erosion, and sequester carbon. Ecological restoration is not feasible because there are only superficial accounts of the composition of the original forests in these areas. In any case, it is difficult to now obtain seed of many of these former species. The restoration design, therefore, involves planting up to five species at each site with no species having more than 70% of the trees. The species being used are those thought to be tolerant of present environmental conditions and most are natives. The local or alpha or diversity is modest but species combinations at each site vary across the province depending on local environmental conditions. Hence, some 40 tree species have been initially identified for use across the region with the possibility that a wider range may be used when further seed sources are identified (in future the trait-based approach for species selection outlined by Giannini et al. 2017 could be useful here). It is too early to assess the successional trajectory of these plantings or their capacity to deliver the ecosystem services required but the managers are confident these multispecies plantings will be more functionally effective than monocultures and more resilient. The wider significance of the project is that it demonstrates the feasibility of undertaking multispecies plantings on such a large scale.

The second example concerns a form of forest restoration being developed in the Philippines to suit the circumstances of small landholders with modest incomes living in largely agricultural landscapes (Nguyen et al. 2014). In this situation, any reforestation must generate a financial benefit and do so relatively quickly. Large individual plantations are not possible but the collective impact of many small plantings can contribute to the restoration of species diversity at a landscape scale and, thereby, help generate various ecosystem services desired by the broader community. The method being developed uses mixtures of fast-growing species that can be harvested after a few years (e.g. for fuelwood) as well as slower-growing species generating other products at successively later stages. Each site may be planted with up to 20 species with the choice of species used at particular sites depending on local conditions and markets. The number of species used is something of a compromise because the species chosen must have some commercial or household value as well as being able to tolerate the environmental conditions at the site. Different farmers will chose different species mixtures meaning that the total number of species introduced across the landscape as a whole will certainly exceed 20 species.

In neither of these two examples did the species' traits entirely dictate species choices. In the China example, the choice was made on the basis of what species were still available and were thought able to now grow at these sites. In the Philippines example, the choice was largely based on the potential commercial values of the various species although environmental tolerances and growth rates were also influential. It is not clear, therefore, how successful these types of mixtures will be in assembling combinations of traits that are able to generate the ecosystem services desired by the broader community. These types of trade-offs deserve further investigation.

Several critiques might be made about these relatively simple restoration designs. The first is that these forms of restoration are too unambitious. But, in both examples, the necessarily modest levels of alpha diversity were also associated with important increases in beta and gamma diversity across the landscape as different species mixtures are used at different sites. It may be that these changes, rather than just improvements in alpha diversity, are critical for improving multifunctionality (Pasari et al. 2013; Thompson & Gonzalez 2016). Furthermore, over time, there is the potential for these plantings to be colonized by additional species dispersed from remnant forest patches (and even other plantings) elsewhere in the landscape (Keenan et al. 1997). More remains to be learned about the functional effectiveness of these different forms of biodiversity at larger spatial scales and over longer time periods (Brose & Hillebrand 2016).

A second concern is that these planting designs assume functionally important biota from other trophic levels will eventually colonize the new forests. As indicated in Table 1, many generalist species may be able to use such simple mixtures but those with more specialized habitat requirements may be unable to do so until the enrichment process described above has modified these habitats. This assumes, of course, that these other biota still remain elsewhere in the landscape and can reach the site.

A final concern about these planting designs is that they might unknowingly include incompatible species and be unstable. This matters where the new forest will not be logged and the intention is to establish a self-sustaining new ecosystem (e.g. the China example). In such cases, the problem will have to be dealt with using adaptive management to encourage the evolution of what Hobbs et al. (2009) have described as novel ecosystems. The problem of potential incompatibilities will be less important when forests are being periodically harvested and replanted (e.g. the Philippines example).

The main implication for both landholders and the global community is that even modest increases in species numbers have the potential to supply a wider variety of ecosystem services (and goods) than industrial timber monocultures and are feasible to use at these landscape scales. These advantages may be enough to persuade many landholders to abandon traditional monocultures in favor of using species mixtures but other incentives, or even legislative obligations, may also be needed. Brancalion et al. (2013) describe how this has been approached in a large-scale restoration program in Brazil.

Scaling Up — Forest Landscape Restoration

Any large-scale forest restoration program is likely to involve a variety of reforestation methods because of the diversity of environmental conditions and landholder aspirations (Stanturf et al. 2014). The problem is how to balance the type, location, and scale of any reforestation in order to maximize the generation of ecosystem services?

An approach known as forest landscape restoration (FLR) is now being developed to achieve this (Lamb 2014; Sabogal et al. 2015). FLR seeks to improve environmental outcomes from undertaking large-scale restoration while also improving the livelihoods of people living in these landscapes. This is conceptually simple but, in practice, may be difficult to implement. At its most basic FLR might involve legally obliging landholders to reforest certain areas such as steep slopes or riverine areas (Rodríguez et al. 2015). Others have approached the issue by developing spatially explicit modeling tools (e.g. Booth 2012; Budiharta et al. 2016; Jones et al. 2016) or planning methodologies (Laestadius et al. 2014) to guide FLR implementation. These types of approaches are useful in providing idealized targets but are mostly dependent on the availability of extensive biophysical data bases and require supportive national planning institutions for their implementation. Neither of these are always available. Furthermore, while some kind of a national planning framework is obviously critical, many decisions must be made at a local level because this is the scale at which decisions can take account of local preferences and make trade-offs in the context of evolving markets for forest products and/or ecosystem services and of the need to intensify agricultural production. What this means in practice is that FLR is likely to involve some "intelligent tinkering" (Mills et al. 2015) or informed "muddling" (Sayer et al. 2008) while appropriate methods and planning institutions are developed.

Under such circumstances it is important to monitor how FLR is carried out and the extent to which ecosystem services are actually generated (recognizing that some may take time to eventuate). Reference or "control" ecosystems are sometimes used in restoration programs and hydrological studies to assess success or measure change. Neither approach is feasible at these large landscape scales, especially when only parts of a landscape are being reforested. This makes it difficult to distinguish signal from noise and means that monitoring will require new methods and metrics to assess FLR success. These metrics should include those measuring changes in the supply of various ecosystem services, changes in the socioeconomic circumstances and restoration objectives of landholders as well as some assessment of the continued effectiveness of the institutional arrangements.

Conclusion

Any kind of large-scale reforestation will be difficult to achieve in practice but there is evidence that restoring forests with only modest levels of species can generate many, though not all, ecosystem services provided species with appropriate traits are used. This should make the task less difficult than if a more ambitious program of ecological restoration was used, particularly if it can be combined with a program of facilitating natural regeneration. But more needs to be learned about the traits needed to generate particular services so that appropriate species assemblages are used rather than simply mixtures of readily available species. Many ecosystem services will also require restoration be done in certain strategically placed locations meaning more also needs to be known about the relationship between the beta and gamma diversity generated by restoration and the generation of ecosystem services.

Increasing forest cover and altering existing land use patterns is likely to be controversial because national goals will not necessarily match the needs and aspirations of local landholders. Future research therefore needs to identify a portfolio of restoration approaches and species assemblages able to meet these various goals. FLR offers the possibility of using these various approaches to satisfy multiple objectives. However, policies and institutions will be needed encourage the use of these new approaches and to facilitate the necessary trade-offs needed to implement them across large landscapes.

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