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Forest restoration following surface mining disturbance: challenges and solutions

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Abstract Many forested landscapes around the world are severely altered during mining for their rich mineral and energy reserves. Herein we provide an overview of the challenges inherent in efforts to restore mined landscapes to functioning forest ecosystems and present a synthesis of recent progress using examples from North America, Europe and Australia. We end with recommendations for further elaboration of the Forestry Reclamation Approach emphasizing: (1) Landform reconstruction modelled on natural systems and creation of topographic heterogeneity at a variety of scales; (2) Use and placement of overburden, capping materials and organic amendments to facilitate soil development processes and create a suitable rooting medium for trees; (3) Alignment of landform, topography, overburden, soil and tree species to create a diversity of target ecosystem types; (4) Combining optimization of stock type and planting techniques with early planting of a diversity of tree species; (5) Encouraging natural regeneration as much as possible; (6) Utilizing direct placement of forest floor material combined with seeding of native species to rapidly re-establish native forest understory vegetation; (7) Selective on-going management to encourage development along the desired successional trajectory.

Jennifer Franklin, Jan Frouz, Sarah Hall, Douglass F. Jacobs and Sylvie Quideau have contributed equally to this manuscript.

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Successful restoration of forest ecosystems after severe mining disturbance will be facilitated by a regulatory framework that acknowledges and accepts variation in objectives and outcomes.

Keywords Mining · Forest restoration · Landform reconstruction · Microbial communities · Soil redevelopment · Tree establishment · Forest understory vegetation

Introduction

A large proportion of the world's mineral and energy resources are found in forested regions, which are consequently subjected to severe disturbance by surface mining. This is a major contributor to the current global need for forest reclamation and restoration (World Resources Institute 2014) and helps explain the increasing interest in forest restoration (Angel et al. 2005; Burton and Macdonald 2011). The broad objective of forest restoration is to return the land to productive capability with an ecosystem comprised of native species and that will function to provide a diversity of economic and ecological values (Grant and Koch 2007; Zipper et al. 2011b). Restoration of forest landscapes after severe mining disturbance presents substantial challenges. For example, the need to re-create the landform complexity that underlies variation in ecosystem structure, composition and function and to redevelop soil types that in natural systems develop slowly over long time periods. Further, forests are structurally complex ecosystems with a diversity of plant species, the dominant ones being very long-lived; thus there is a need to account for long-term successional development of these ecosystems. Finally, there is the need to ensure the resilience of reclaimed ecosystems in the face of future natural and anthropogenic disturbances.

Early efforts to reclaim forested lands after mining disturbance often focused on revegetation with relatively little attention to landform re-creation and the use of native species or re-establishment of tree cover (Grant and Koch 2007; Gorman et al. 2001; Skousen and Zipper 2014; Zeleznik and Skousen 1996). This has changed in recent years, however. For example, the Surface Mining Control and Reclamation Act (SMCRA), enacted in the United States in 1977 caused a major change in reclamation practice by requiring assessment of the characteristics of the potential mined site and an accompanying strategy to restore environmental conditions after mining (Skousen and Zipper 2014; Zipper 2000). The initial focus was on reducing erosion; this led authorities to encourage grading and smoothing of reclaimed land surfaces and rapid establishment of grasses and legumes, often agricultural forages (Chaney et al. 1995; Plass 1982; Torbert and Burger 2000). If these lands were not used for grazing, however, they could naturally revert to woody vegetation, particularly on sites adjacent to forest (Franklin et al. 2012; Skousen et al. 2006). Alternatively, sites would often have compacted soils with chemical properties quite different from native soils; this, plus the dense cover of non-native herbaceous vegetation, inhibited establishment of forest cover for decades (Angel et al. 2005; Zipper et al. 2011a). The Forestry Reclamation Approach (FRA) was developed to address this challenge in the Eastern USA coal region, which is comprised primarily of temperate deciduous forest (Burger et al. 2005a; Zipper et al. 2011b). It described an approach to reclaiming mines to forest vegetation that considered creation of a rooting medium and rough surface, thinking these would be good for tree growth and would encourage recolonization of native species. Further it emphasized using ground cover species that would have fewer negative competitive effects on trees and it promoted good planting techniques with a diversity of tree species. Finally, the FRA focused on setting up conditions to encourage future natural plant community succession.

In Alberta, Canada, the rapidly expanding oil sands mining industry is located in the boreal forest, another rich and diverse forest ecosystem being severely impacted by mineral extraction. Reclamation regulations have changed dramatically over the past five decades (Powter et al. 2012). The Alberta Environmental Protection and Enhancement Act, which currently governs conservation and reclamation of mined lands in Alberta, includes the stated objective to "return the specific land to equivalent land capability" (Government of Alberta 2014). Approvals for oil sands mines require operators to "reclaim the land so that the reclaimed soils and landforms are capable of supporting a self-sustaining, locally common boreal forest….integrated with the surrounding area" (e.g., Alberta Environment Approval #46586-00-00 Imperial Oil Resources, Kearl Oil Sands Processing Plant and Mine). A number of regulations within mine operating approvals are designed to help meet this objective. For example, there are requirements for salvaging, stockpiling, and replacing soil materials, including all organic matter on the surface of the soil; a particular emphasis of this is the use of upland surface soils. Other supporting guidelines have also been developed including a classification system for determining land capability (Cumulative Environmental Management Association 2006), detailed guidelines for reclamation that focus on recreating a diversity of ecosite types in which landform, soils and vegetation are compatible (Cumulative Environmental Management Association 2009), and a criteria and indicators framework for assessing success (Poscente and Charette 2012).

There is a long tradition of post-mining reclamation in European countries. For example, with the general mining law #146 of 1854, the Emperor of the Austrian Hungarian Empire established an obligation to return mined landscapes to their original use by reclamation. This law became the basis for mining and reclamation in most successor countries of the Empire. Most European countries now have in place obligations for mining companies to restore mined lands to their original land uses and require that some reclamation planning be undertaken before mining starts. The particular reclamation techniques in Europe are variable; recently some legal regulations recognized the value of near natural processes to restore diversity and protect rare species (Hu 2014). Regulations in other jurisdictions are variable and in some regions there are relatively few regulations in place to ensure restoration of forest ecosystems after severe industrial disturbance.

Objectives for restoration of jarrah forest sites following bauxite mining in Australia include protecting native biodiversity, water catchment protection, timber productivity and rebuilding forest ecosystems that are compatible with values and uses of the surrounding forest landscape (Gardner and Bell 2007; Grant and Koch 2007). Current practices for restoration post-bauxite mining are a product of continuing evolution of practices and regulations, informed by research and driven by intense public scrutiny (Grant and Koch 2007; Koch and Hobbs 2007). The initial practice of planting exotic pines or non-native eucalypts was replaced in the late 1980's by use of only native tree species and re-establishment of native understory vegetation by sowing, planting or direct placement of surface soil materials (Grant and Koch 2007; Norman et al. 2006). At the same time, re-contouring fashioned after natural landforms and deep ripping of the mine pit floor were introduced (Gardner and Bell 2007). An understanding of the resilience of unmined forests to natural disturbance underpins restoration practices and guidelines (Grant and Koch 2007).

Herein we review the current state of knowledge regarding reclamation and restoration of forest ecosystems following surface mining disturbance, explore common challenges, and synthesize the principles that could underlie development of best practices. We draw upon examples from different regions (mainly North America but including some from Europe and Australia) each of which has valuable knowledge to contribute. We conclude by suggesting a further evolution of the Forestry Reclamation Approach (FRA) that can be applied broadly to meet objectives for forest restoration.

Topographic variation from microsite to landscape

The first important consideration of land reclamation is the reconstruction of the disturbed landform, which is ideally guided by the pre-existing landform and projected post-mining land use. Topographic reconstruction is a critical step in the reclamation process, as the rebuilt landform is the foundation for all following reclamation practices (Toy and Chuse 2005). The challenge of topographic reconstruction is to produce landforms that approximate a state of dynamic equilibrium under prevailing environmental and climatic conditions (Toy and Black 2000) and that meet the objectives for post-mining land uses (Skousen and Zipper 2014). At the landscape scale, consideration should be given to the type and arrangement of landforms along with the placement, stability, hydrological and chemical characteristics of the material types used to construct them (Fig. 1). To develop a functional landscape, ponds, lakes, and streams must be reconstructed and located in such a way as to tie them into the geohydrological setting; this should be similar to natural systems where these components are hydrologically connected at a range of spatial and temporal scales (Fig. 1; Devito et al. 2012).

Rebuilt landscapes need to be in a state of dynamic equilibrium so that they are able to develop and evolve, and can be sustained through effects of climate (e.g. precipitation and storm events), changes in topography (e.g. land consolidation and subsidence), and properties of the geological material types (e.g. hydrological and chemical properties, erosion risk, slope stability) (Devito et al. 2012). Climate cannot be controlled but water availability and redistribution, as well evapotranspiration demand, can be managed through geology, cover soils, and re-established vegetation, including their spatial arrangement and connectivity in the rebuilt landscape. When water is in excess, the design of these landscapes and landforms should allow for the redistribution of water to either storage, to be available during extended dry periods, or to other locations in the landscape. Devito et al. (2012) suggested that the heterogeneity of geological materials and topography should be used to create landforms with ample variation at the meso- and micro-scale, which will in turn influence water storage and movement.

The landscape design is the foundation upon which the cover soils and the subsequent ecosystems are placed. Therefore significant consideration needs to be given to cover material types, their placement and arrangements, and the surface conditions that will influence early ecosystem development as well as future trajectories. Layering and thickness of the surface soil materials as well as meso- and micro-topographic variation are key to the re-establishment and sustainability of forest ecosystems and plant communities (Figs. 1, 2). At these scales, topography and habitat heterogeneity appear to be important variables for the establishment of both plants and associated invertebrate communities (Tropek et al. 2013). Mesotopographical variation (1–10 m scale) increases capture (larger surface area) and longer-term storage of moisture (Toy and Black 2000). Creating

Fig. 1 Landform reconstruction, showing heterogeneity of materials and meso-topography, before (*above*) and after (*below*) revegetation after oil sands mining in Alberta, Canada. Photo courtesy of Syncrude Canada Ltd

undulating or hilly surfaces allows for the development of structures such as shallow depressions (swales) designed to collect or potentially redirect water; these can be used to collect and hold water and are most useful in reforestation of degraded, mostly-bare, arid or semi-arid hillsides where water can be directed to trees and vegetation (Toy and Chuse 2005). These structures are most useful when they follow contour lines, as water will otherwise not flow or will flow only over short distances (Tropek et al. 2013).

Layering of cover material types with different textures creates interfaces that can influence moisture availability and field capacity of soils (Zettl et al. 2011). For example downward water movement in layered soil materials can be influenced by capillary barriers that form when unsaturated fine materials overlay unsaturated coarser materials (Miyazaki et al. 1993; Alfnes et al. 2004; Si et al. 2011) or by hydraulic barriers that are due to the presence of a fine, less permeable layer underlying coarser-textured soil (Hillel and Talpaz 1977; Si et al. 2011). Engineered soil covers often include capillary barriers created by the layering of finer textured soil over coarser textured soil (Stormont 1996). The presence of a

Fig. 2 Soil replacement following coal mining on a reclamation site in southwestern Indiana, USA. Operational procedures in this region call for level grading of the replacement soil followed by planting of native hardwood trees. Photo credit: D.F. Jacobs

capillary barrier can increase the available water holding capacity and thus the water available for plant growth, lateral drainage, and/or percolation (Huang et al. 2013). This in turn will influence plant species composition, root distribution, leaf area development, and overall productivity of reclaimed sites. An important consideration in placing soil materials is to avoid compaction, which can hamper planting, reduce soil air and water holding capacity, and impede root growth (Grant and Koch 2007; Sweigard et al. 2007).

Micro-topographic and soil heterogeneity can be advantageous for mine reclamation (Kappes et al. 2012). Micro-topography has a critical influence on reclamation success and should be aligned with the objectives for post-mining land use, topography, and drainage systems. A rough surface can facilitate natural revegetation and encourage plant species diversity (Groninger et al. 2007; Mackenzie and Naeth 2010; Skousen et al. 2006; Schott et al. 2014). Microtopographical variation increases the range of environmental conditions allowing for a greater variety of species to establish. Surface roughness produces variation in conditions at the 1 cm to 1 m spatial scale, in turn creating wide variation in moisture availability and exposure to sun and wind. On reclaimed sites in the boreal forest, small-scale variation and heterogeneity in cover soil materials and their nutrient and carbon status appear to play a role in seedling establishment and early success (Wolken et al. 2010; Pinno et al. 2012, 2014). Surface roughness can be achieved by reduced smoothing and "treading in" of the surfaces during the reconstruction process, or can be artificially increased through the deposition of materials such as woody debris on the surface (Grant and Koch 2007).

Woody debris can be beneficial during reclamation for a variety of reasons. When placed on a mine reclamation site, woody debris can influence soil microbial communities, improve soil nutrient and water holding capacity, and can contribute to development of nutrient cycling (Brown and Naeth 2014; Dimitriu et al. 2010; Skousen et al. 2011). Large woody debris can ameliorate soil temperature and moisture extremes through provision of shelter (Haskell et al. 2012) and also increases surface roughness, providing microsites for vegetation establishment. Interestingly, recent research suggests that inorganic and inert substrates such as bricks could also serve this purpose (Bijman, Landhäusser and

Macdonald, unpublished). Large woody debris can also be used to restrict access by humans or other animals, such as predators, to a reclaimed site. Wood chips and mulch have also been explored as reclamation amendments (McConkey et al. 2012). For example application of a woody mulch covering to unweathered gray sandstone materials improved tree growth to the level observed in brown sandstone materials (Wilson-Kokes et al. 2013b). However, these types of finer woody debris can insulate the ground (creating late thaw), alter nutrient dynamics, and impede emergence and establishment of vegetation (Landhäusser et al. 2007; Vinge and Pyper 2012) and thus their use should be confined to shallow depths and for specific objectives.

Some regulatory frameworks are, to some extent, supportive of practices aimed at creation of topographic heterogeneity at a variety of scales. For example, a key provision of the SMCRA for the mountainous Appalachian coal region in the USA is landform reconstruction to "approximate original contour" and restoration of landform stability. "Approximate" was used because Appalachian rocks "swell" from 10 to 30 % after blasting so the resulting landform may have slightly different forms and elevations than what existed prior to mining (Skousen and Zipper 2014). State regulations in the USA generally require landform stability and this is based on slope steepness and length, material properties such as particle size and texture, as well as compaction criteria. Water management is based on performance standards for water quality, stream shape and size, storm-water management, and pond number and size relative to the size of the disturbed area.

In the oil sands mining region of Alberta, Canada, landform reconstruction is based on creating landforms that meet the criteria for geotechnical stability while also functioning to regulate water flow through the landscape (Devito et al. 2012). There is a need to ensure adequate supply of moisture to support growth of forest vegetation across a diversity of topographic positions and aspects while at the same time less suitable overburden material (e.g., saline-sodic materials) might need to be spatially separated from the rooting zone. Topographic heterogeneity is modelled on the natural landscape, and relationships of forest ecosite types with topography and soils are used as a guide (e.g., Beckingham and Archibald 1996). Landforms are constructed with relatively mild topographic relief with the aim to establish a diversity of forest ecosystem types on the upper positions, and wetlands in the lower topographic positions (Rowland et al. 2009).

Soil establishment and development

The placement of a suitable rooting medium is a critical element in reclaiming surface mined lands, as soil serves as the foundation of the new ecosystem (Zipper et al. 2013). Soils developing on reclaimed sites are highly modified, as compared to native soils, in terms of their physical, chemical, and biological properties and their vertical arrangement into distinct horizons; indeed a new classification system has been developed for such soils, termed Anthroposols in Canada (Naeth et al. 2012), Udorthents in the USA (Sencindiver and Ammons 2000), and Technosols in the world reference base for soil resources (IUSS Working Group WRB 2006). Regulations in many jurisdictions require the salvage and stockpiling of topsoil for later use during reclamation; e.g., the SMCRA in the USA, EPEA approvals of oil sands mines in Alberta, Canada, ministerial approvals for bauxite mining in Western Australia. In many European countries topsoil salvage is only required when reclaiming to agricultural land. Native topsoil provides organic matter, soil fauna and microorganisms, native plant propagules and helps maintain nutrient capital on the site thus facilitating re-establishment of nutrient cycling (Grant et al. 2007; Skousen et al. 2011; Mackenzie and Naeth 2010; Macdonald et al. 2015). However the nutrient and biological quality of topsoil decreases during stockpiling (Abdul-Kareem and McRae 1984; Grant et al. 2007; Mackenzie and Naeth 2010). Salvaged top soil is a valuable "living resource" and it should be used/redistributed as soon as possible after salvage (Grant and Koch 2007; Koch 2007; Macdonald et al. 2015; Zipper et al. 2013).

Importance of the soil organic component

Because of the proportionally higher surficial versus belowground litter additions in forest soils a significant fraction of the total ecosystem carbon often accumulates on top of the mineral soil to form several partially decomposed organic horizons, collectively defined as the forest floor. This is particularly true of boreal forests, where the forest floor alone may contain as much carbon as the entire aboveground vegetation (Goodale et al. 2002), representing a storehouse of nutrients and a major determinant of biogeochemical fluxes (Harden et al. 1997; Prescott et al. 2000). Vegetation type also has a direct bearing on soil carbon accumulation; for example boreal coniferous forest stands showed greater forest floor biomass but higher mineral soil carbon than deciduous or mixed stands (Kishchuk et al. 2014; Laganière et al. 2013). In contrast, warmer, temperate and tropical forests have minimal forest floor accumulation, but relatively more organic matter is typically found deeper in the mineral soil (Vogt et al. 1995). These clear differences among ecosystems should be carefully considered when reconstructing forest soils and re-establishing nutrient stocks so as to ensure that restoration practices are tailor-made for each ecosystem type and its associated climatic conditions.

Soil carbon is often used as a proxy for soil quality and is a good indicator of ecosystem recovery in post-mining soils (Frouz et al. 2009; Turcotte et al. 2009). Results from studies using chronosequences post-fire and post-harvesting indicate that soil carbon may return to pre-disturbance levels after about three decades in boreal forests (Norris et al. 2009), while recovery of soil carbon in secondary tropical forests takes between 20 and 100 years (Martin et al. 2013). In post-mining soils where no organic amendment is used, the potential for carbon accumulation is large (Akala and Lal 2000), although carbon levels may not recover to those of natural soils, even after several decades (Zipper et al. 2013). A meta-analysis of carbon storage during post-mining soil development showed that most sites reached the pre-mining level of soil organic carbon within 20 years. However, vegetation types (grassland, deciduous forest, coniferous forest) showed differences in the temporal trend in rate of organic matter accumulation, effects of temperature, and where the organic matter was stored (mineral soil versus litter, fermenting litter and humus layers) (Vindušková and Frouz 2013). Overall, the best results in terms of organic matter accumulation seemed to be achieved under the natural vegetation type of a given biome (Vindušková and Frouz 2013).

Whether it is native to the soil or added as an amendment, organic matter can improve many if not all of the key chemical and physical characteristics of forest soils, including its total nutrient stocks, its aggregation, and its water holding capacity (Larney and Angers 2012; Quideau et al. 2013a). In terms of soil biological properties, however, organic amendments (even those derived from local topsoil) may not provide the same advantage as native organic matter (Vetterlein and Hüttl 1999). Clear inter-relationships are apparent between vegetation, soil organic matter, and soil microbial communities of natural forests (Merilä et al. 2010) but these linkages may be disrupted in post-mining soils, even when organic amendments are applied (Quideau et al. 2013b). For instance, a measurable evolution of organic matter composition with time was reported in chronosequences of reconstructed boreal soils (Sorenson et al. 2011), but parallel changes in the soil microbial communities were much harder to establish (Dimitriu et al. 2010; Insam and Domnsch 1988). The main biological benefit of organic amendments may be to promote plant growth—hence indirectly, rather than directly, affecting soil biodiversity (Hahn and Quideau 2013). Ongoing advances in our fundamental understanding of the complex interactions between plants, microorganisms, and soil carbon processes (e.g., Averill et al. 2014) should provide some of the answers needed to improve use of organic amendments during forest restoration.

Soil microbial processes

Physical, chemical, and biological components of soil play important roles in the development of functioning ecosystems on severely disturbed lands and these different factors are not isolated, but rather interact with one another in important ways (Rowland et al. 2009). Re-establishment of decomposition and nutrient cycling processes is an essential aspect of post-mining forest restoration (Grant et al. 2007). Microorganisms are essential for nutrient cycling, especially N and C (Mummey et al. 2002; Ingram et al. 2005; Littlefield et al. 2013), on severely disturbed mine sites. They also have important beneficial and pathogenic interactions with plants (e.g., Grant and Koch 2007; Jasper 2007).

Soil bacteria typically fall into the r-strategists category of organisms adapted to take advantage of resource changes and disturbances while fungi are more typically slower growing and more conservative K-strategists (Sylvia et al. 2005). Thus, undisturbed forest soils tend to be dominated by fungi, whereas forest soils that have been recently disturbed have lower numbers of fungi and higher numbers of bacteria (Dimitriu et al. 2010). Mummey et al. (2002) found that fungal biomarkers were highest under plant canopies and were also positively correlated with soil organic matter.

Vegetation composition affects organic matter content and quality on reclaimed mine sites, in turn affecting the composition of microbial communities (Mummey et al. 2002; Grant et al. 2007; Frouz et al. 2009; Sorenson et al. 2011; Macdonald et al. 2012). Sorenson et al. (2011) found that coniferous stands were linked with fungal presence, whereas aspen stands had a stronger bacterial component. The type of soil replacement or substrate amendment can be clearly linked with differences in both vegetation (Angel et al. 2008; Miller et al. 2012; Wilson-Kokes et al. 2013a) and microbial biomass and composition (Machulla et al. 2005; Palumbo et al. 2004; Chodak and Niklinska 2010). Vegetation affects soil microbial communities by the quantity and quality of litter while microbes, in turn, influence re-vegetation success (Jasper 2007).

Many plant species require microbial symbionts for their germination and growth. For example, at least 75 % of native plant species in jarrah forests form mycorrhizal relationships (Jasper 2007). Use of, or amendment with, topsoil or O-horizon material can introduce both soil microorganisms and plant propagules (Koch 2007; Hall et al. 2010; Mackenzie and Naeth 2010; Zipper et al. 2013). Subsequently, litter layer development is key to establishment of diverse and abundant mycorrhizal communities (Jasper 2007). On restored bauxite mines in Australia, mycorrhizal communities can recover to those in unmined jarrah forest within 10–15 years when soil handling practices promote survival of microorganisms and there are nearby unmined forests that can serve as source populations (Jasper 2007).

On the other hand, some microorganisms act as pathogens, and in this case mine spoils may offer a more hospitable (i.e., pathogen-free) medium than would forest soils that have not been subjected to severe disturbance. For example, the potential of the American chestnut (*Castanea dentata* (Marshall) Borkh.) to thrive on surface-mined lands is due in part to the lack of pathogenic *Phytophthora* in newly created soils (French et al. 2007; Jacobs 2007; Jacobs et al. 2013).

Soil fauna

The soil micro-flora plays a crucial role in the decomposition of plant litter (Anderson and Ineson 1984; Lavelle et al. 1997). While the soil fauna contributes little to plant litter mineralization, it does affect formation of various biogenic structures such as pores and aggregates, and influences the distribution of organic matter in the soil profile (Lavelle et al. 1997; Ponge 2003; Frouz 2013; Frouz et al. 2013a). This, in turn, affects soil characteristics and processes such as porosity, infiltration, water holding capacity, microbial biomass, fungal:bacterial ratio and carbon storage (Frouz et al. 2013a, b, d; Frouz and Kuráž 2013).

Colonization of post-mining soils by soil biota will be affected by fauna available in the surrounding landscape, the substrate and vegetation of the post-mining site, distance from an undisturbed ecosystem, and migration barriers such as mines or unsuitable vegetation (e.g., Majer et al. 2007). Generally, dry grassland soils are dominated by root feeding soil fauna while wetter forest soils are dominated by a saprophagous fauna (Frouz et al. 2013b). What is a good migration corridor for one group may be a barrier for another. For example, ground crawling saprophagous macrofauna may migrate well in dense forest patches but such sites can be a barrier for some flying arthropods (Frouz et al. 2013b).

Litter properties, in particular the C:N ratio, will strongly affect humus form development and, in turn, soil fauna composition of the reclaimed site. Soil under trees producing slowly-decomposing litter with a high C:N ratio, high tannin content, and wax cuticle, is colonized mainly by soil meso- and micro-fauna which contribute little to bioturbation and mixing of organic matter and mineral soil (Frouz et al. 2001; Ponge 2003). As a consequence, a mor type of humus with a thick fermentation (Oe) layer can develop (Ponge 2003). This, in turn, supports high fungal biomass and high density of soil meso- and micro-fauna (Frouz et al. 2001, 2013b, d). Studies in the Czech Republic showed that mesofauna density on reclaimed sites can reach densities comparable to undisturbed forest within 15–20 years (Frouz et al. 2013c). On the other hand, in forests receiving litter with a lower C:N ratio soil macrofauna activity contributes to litter fragmentation, mixing and bioturbation leading to development of a moder or even moder-mull type humus (Ponge 2003). This is associated with higher microbial biomass, and often with higher soil carbon (Frouz et al. 2013d).

Re-establishment of vegetation

A complex above- and below-ground vertical stratification characterizes most natural closed-canopy forest ecosystems and reconstructing this state after resource extraction is a fundamental objective in forest reclamation and restoration (Gorman et al. 2001; Skousen et al. 2006; Grant and Koch 2007; Macdonald et al. 2012). The eventual plant community that develops on reclaimed mine lands will be a function of: landform and topography, local climate/weather patterns, soil properties, vegetation re-establishment treatments, subsequent disturbance, wildlife impacts, and land management (e.g., Angel et al. 2008;

González-Alday et al. 2008; Burger and Fannon 2009; Tropek et al. 2013). Re-establishment of a closed canopy consisting of native tree species can help promote establishment of a diverse native understory plant community (e.g., Parrotta et al. 1997; Koch 2007). Further, vegetation re-establishment is critical for rebuilding nutrient capital, facilitating soil development, and supporting re-colonization by native vertebrate and invertebrate fauna (Majer et al. 2007; Grant et al. 2007; Nichols and Grant 2007).

Creating the conditions for vegetation establishment

When choosing species for reclamation, it is important to consider site characteristics including available moisture, soil texture, salinity and pH, soil fertility and nutrient availability (Purdy et al. 2005; Moreno-de Las Heras et al. 2008; Zipper et al. 2011a; Davis et al. 2012). Soil moisture availability is particularly important for vegetation development (Huang et al. 2013). In the northern hemisphere, the northern aspect of hillsides often has greater soil moisture, which can result in more rapid establishment of vegetation (González-Alday et al. 2008). Coarsely textured soils on upper slopes often have very low available soil moisture, even in moist climates, and so require drought tolerant species (Zipper et al. 2011a). Excessive soil moisture can also occur on mine reclamation sites, often in seeps on the lower half of slopes or at the surface of areas with heavily compacted soils where water cannot infiltrate into the soil (e.g., Devito et al. 2012). Understory species classified as facultative wetland species are good choices for these areas, and for areas of compacted material where low soil aeration favors flood-tolerant species (Moreno-de Las Heras et al. 2008).

Soil chemical properties will influence re-vegetation success. Soil liming and fertilization can make mine soils more suitable for vegetation establishment if conditions are limiting (Grant et al. 2007; Skousen and Zipper 2010) although fertilization can lead to increased cover and richness of exotic species (Norman et al. 2006). Mine soils with high levels of heavy metals can be treated to reduce their bioavailability or metal-tolerant plant species can be chosen for revegetation of such sites, but it is important to consider whether these species accumulate metals in foliage (Wong 2003). Similarly, saline or sodic spoils can be treated but may also require the selection of salt tolerant species, which may be available from naturally saline sites within the mining region (Purdy et al. 2005). If the pH of the reclaimed site differs from that of the surrounding areas it will be particularly important to actively establish site-specific vegetation because there may not be suitable species available to colonize the site from adjacent areas and natural succession will proceed slowly (Skousen et al. 1994; Moreno-de Las Heras et al. 2008).

The nature of the overburden material has an important influence on the physical and chemical properties of the developing soil, in turn influencing re-vegetation success. This is particularly well illustrated by reclamation of coal mining sites in the eastern USA. Early reclamation efforts (1930s–1970s) involved planting of trees into loose, uncompacted weathered overburden (Potter et al. 1955; Zeleznik and Skousen 1996; Gorman et al. 2001; Ashby 1998). These sites showed good tree growth and rapid recruitment of adjacent native trees resulting in diverse forest after 10–20 years (Skousen et al. 1994, 2006; Tryon 1952). Later the influence of mine soil properties on tree growth was explored in more detail (Andrews et al. 1992; Ashby 1998; Daniels and Amos 1985; Johnson and Skousen 1995; Rodrigue and Burger 2004; Torbert et al. 1990; Zeleznik and Skousen 1996).

While the SMCRA now requires salvage and reuse of topsoil, use of substitute topsoil materials is often permitted. Overburden rock types that are used as “topsoil” include weathered and unweathered sandstones, siltstones, and shales (Smith and Sobek 1978). When placed on the surface these materials will weather and transform over time into a variety of mine soil materials with vastly different physical and chemical properties (Sencindiver and Ammons 2000). Overburden materials have lower nutrients than native topsoils (Sobek et al. 2000) increasing the need for use of fertilizers and mulches (Mays et al. 2000). Rock spoils are also initially devoid of pedogenic organic C (Daniels and Amos 1985) and it will accumulate only slowly on re-vegetated mine soils (Amichev et al. 2008).

Weathered sandstones produce soils with a sandy loam texture and slightly acidic pH (4.5–6.0), both of which are conducive to tree growth (Torbert and Burger 2000; Burger et al. 2005b; Skousen et al. 2006). Siltstone and shale materials weather rapidly but the resulting soils have finer textures and can thus be prone to compaction or settling (Casselman et al. 2006). Unweathered gray sandstone, siltstone and shale materials result in soils with slightly higher pH (from 6.5 to 8.5), in which trees can survive but grow slowly (Angel et al. 2008; Miller et al. 2012). Soluble salts, which are detrimental to tree growth, are higher in fine textured soils derived from unweathered, shale and siltstone overburden as compared to soils derived from weathered sandstone (Torbert et al. 1990; Rodrigue and Burger 2004).

Overall, longer-term survival and growth of trees is better in weathered brown materials than in unweathered gray sandstone, siltstone, shale (Angel et al. 2008; Emerson et al. 2009; Wilson-Kokes et al. 2013a) (Table 1). This has been attributed to a slightly acid pH, low soluble salts, good drainage and aeration, and better water retention compared to most unweathered rock materials. However gray unweathered sandstone can undergo rapid physical and chemical change (Haering et al. 2004) such that soil development can begin within 3 years (Sencindiver and Ammons 2000; Bendfeldt et al. 2001). Further, unweathered rocks generally contain more base cations (Zipper et al. 2013; Wilson-Kokes et al. 2013a) and these are especially important to species such as tulip poplar, ash, maples, and some oak species (Rodrigue and Burger 2004; Burger and Fannon 2009). When rock spoils have aged sufficiently to leach soluble salts and stabilize in pH (i.e., ~10–20 year), soils with pHs in the slightly acid to circumneutral range can in some cases be more productive for native hardwoods than more acidic soils (Rodrigue and Burger 2004; Burger and Fannon 2009). A mix of native top soil, weathered sandstone and unweathered rock materials could result in optimal mine soil quality for restoring native forest productivity in the long term (Burger and Fannon 2009; Rodrigue and Burger 2004).

Table 1 Volume (height × diameter^2 in cm^3) of 10 planted tree species on topsoils derived from weathered brown versus unweathered grey overburden 3- and 8- years post-establishment at Catenary Coal Mine in Kanawha County, West Virginia

Species	Treatments			
	3 years		8 years	
	Brown	Gray	Brown	Gray
Black cherry	523	155	930	1981
Black locust	1294	115	5324	328
Dogwood	209	19	2534	1064
Redbud	124	84	2058	1762
Red oak	132	32	3609	397
Sugar maple	44	36	354	108
Tulip poplar	591	59	3910	284
White ash	204	33	3500	711
White oak	108	38	3027	74
White pine	34	10	762	154
Mean	308	55	3913	449

There was a significant difference between the two soil types in each time period

On reclaimed sites in the boreal forest, cover soil material and their nutrient and carbon status appear to play a role in determining revegetation success (Wolken et al. 2010; Pinno et al. 2012, 2014). On naturally saline sites in the boreal forest, Lilles et al. (2010, 2012) found that forest vegetation could establish on sites with extreme values of salinity and high pH as long as these were confined to the lower soil profile (40 cm or deeper). White spruce (*Picea glauca* (Moench) Voss) grew slowly on saline sites but was able to survive and form good forest cover (Lilles et al. 2012). Productivity of trembling aspen was quite good on saline sites that had good soil moisture and nutrient availability but there was some evidence of declining productivity over time, which could be related to restrictions on the suitable rooting zone (Lilles et al. 2012).

Planting trees and shrubs

The selection of tree species for a reclamation site should consider the influence of landform, topography, local climate/weather, and soils. As soil conditions on reclamation sites are often highly variable, planting a mixture of tree species is often advisable; this also provides of a variety of potential habitats and allows the developing forest to build resistance and resilience to pests and other stressors (Parrotta et al. 1997; Davis et al. 2012). Deploying tree species representing different life histories, tolerances to stress, and successional status can have other benefits, such as increases in forest productivity through facilitation and resource partitioning, as well as increases in diversity of associated above- and below-ground flora and fauna. For example, when three upland boreal tree species were planted in salvaged lowland peat, upland mineral forest floor, and a sub-soil, initial soil mycorrhizal diversity was driven by tree species and not the substrate type (Fig. 3).

On most forest reclamation sites, the use of direct seeding or a reliance on natural seed rain for tree regeneration is generally avoided because tree establishment tends to be slow and irregular and sites tend to become dominated by less desirable, wind-dispersed species. Tree planting, therefore, remains one of the most effective strategies in re-establishing forest cover in areas affected by industrial disturbance (e.g., Parrotta et al. 1997). Good progress has been achieved, however, with re-establishment of native jarrah forest following bauxite mining in Australia through sowing of diverse native tree seed mixes and use of surface soil materials (Grant and Koch 2007). One aim of rapidly establishing trees on a reclamation site is to quickly create a continuous tree canopy in order to suppress the establishment of shade intolerant, "weedy" herbs and shrubs that could hinder the development of native forest understory vegetation.

The target seedling concept is based on "fitness for purpose" in which limiting factors of reclamation sites are used to determine ideal seedling stock types, with species- and site-specific considerations. Access to high quality nursery planting stock helps to promote and accelerate forest reclamation, as environmental conditions on reclamation sites are generally much more stressful than those characterizing typical reforestation sites (Davis et al. 2010). However, much of the knowledge regarding seedling production techniques is based on commercially important tree species used for reforestation (Oliet and Jacobs 2012). Nursery stock quality specifications for the diversity of tree species desired for reclamation is still in development or in some cases unavailable.

Seedling physiological and morphological characteristics can be manipulated during nursery culture conferring an advantage on stressful reclamation sites. For example high tissue C reserves in aspen (*Populus tremuloides* Michx.) seedlings positively affect field performance (Landhäusser et al. 2012b). On nutrient-limited sites, nutrient loading during the nursery phase allows for luxury consumption of nutrients in plant tissues (Birge et al.

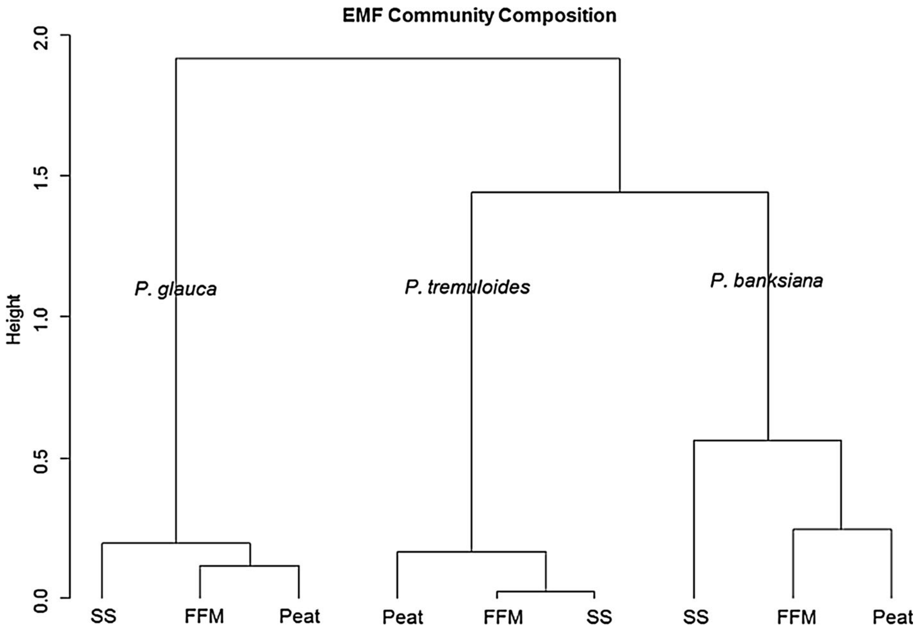

Fig. 3 Results of cluster analysis of ectomycorrhizal fungal (EMF) communities showing the influence of tree species (white spruce (*Picea glauca)*, trembling aspen (*Populus tremuloides*); jack pine (*Pinus banksiana*)) and three different reclamation soil capping substrates (*SS* sub-soil, *FFM* forest floor material, *Peat* peat)

2006; Salifu and Jacobs 2006), which may then be re-translocated to promote growth following field planting (Salifu et al. 2009; Schott et al. 2013). On exposed upland reclamation sites where soil moisture is often limiting, seedlings with high root-shoot ratios should be better adapted to deal with the evaporative demand. However, there are often trade-offs between physiological and morphological characteristics such as tissue storage of nutrient and total C reserves and the balance of root-shoot and total seedling size (Villar-Salvador et al. 2012). For example, tall and large trembling aspen seedlings that had high total nonstructural carbon reserve but lower tissue concentration of reserves, performed more poorly after outplanting than did a smaller stock type with higher tissue concentrations and root to shoot ratios (Landhäusser et al. 2012a; Fig. 4). In contrast, on the same

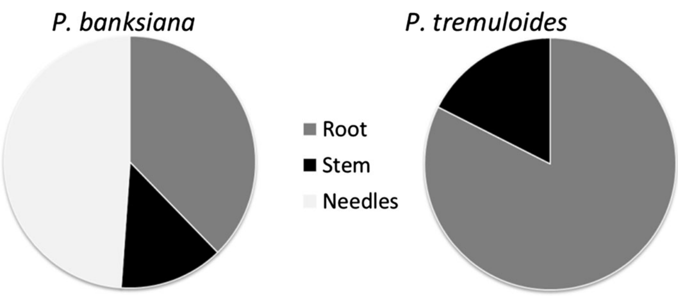

Fig. 4 Distribution of tissue carbon reserves in organs of dormant jack pine (*Pinus banksiana*) and trembling aspen (*Populus tremuloides*) seedling stock prior to outplanting

site large jack pine (*Pinus banksiana* Lamb.) seedlings with high total carbon reserves performed better after outplanting, while tissue reserve concentrations did not play a role (Goeppel, Landhäusser et al. unpublished data). These differences are likely driven by the relative location of reserves in the seedlings. While the majority of carbon reserves are stored in the needles of jack pine seedlings, in aspen seedlings the majority of reserves are stored in the roots (Goeppel et al. unpublished).

In addition to seedling stock characteristics, site and substrate conditions can be manipulated during reclamation to reduce stress on the outplanted seedlings (Franklin et al. 2012; Löf et al. 2012). Other factors such as seedling handling (Davis et al. 2010), planting procedures, timing of planting (Landhäusser et al. 2012b), and microsite/planting spot selection will also influence the successful establishment and growth of tree seedlings on reclamation sites.

Natural regeneration

In some circumstances, natural regeneration in post-mining sites can result in development of forest stands (Frouz 2013; Fig. 5). Forest stands naturally establishing on reclamation sites are typically formed by tree species with air borne seeds such as *Populus* spp. and *Betula* spp. (Frouz et al. 2013c). Grading of the surface or application of topsoil can actually reduce natural regeneration of trees, increase compaction (inhibiting tree rooting) and increase the abundance of competing ground vegetation (Ashby 1998; Frouz 2013; Skousen et al. 2011; Zeleznik and Skousen 1996). Deep ripping can be used to improve compacted soils (Grant and Koch 2007). Successful natural establishment of trees by seed is facilitated by the presence of an ungraded rough surface that can trap seeds and serve as a favorable regeneration microsite, partly because these will tend to host less competitive vegetation (Frouz 2013; Frouz et al. 2011; Landhäusser et al. 2010). Natural regeneration of trembling aspen on reclaimed mine sites was facilitated by concave microsites and mixed mineral-organic substrates (Schott et al. 2014).

Natural regeneration can result in different tree composition and lower woody biomass than in planted forest (Frouz et al. 2013c; Skousen et al. 1994) but these differences are most pronounced in the early stages of development (e.g., in the Czech Republic, first 20 years) (Frouz 2013). Indeed, natural regeneration can sometimes result in greater species diversity and more rapid re-development towards natural vegetation composition than active reclamation by planting of trees; this is particularly true if the latter involves planting of non-native species and/or re-contouring the site and causing disturbance to naturally establishing vegetation (Hall et al. 2010; Hodačová and Prach 2003). Natural regeneration can be less costly and can result in a forest stand comprised of locally adapted species and genotypes resulting in greater floral and faunal diversity than actively reclaimed sites (Prach et al. 2001; Frouz 2013; Skousen et al. 2006). On the other hand, the outcomes of natural regeneration are less predictable and generally much slower. Establishment of forest on coal mine lands in the eastern USA by means of natural regeneration was extremely slow, even where soils were favorable (Zipper et al. 2011a). This can be attributed to establishment of non-native invasive plant species (Zipper et al. 2011a) and "arrested" development of vegetation succession on reclaimed land with compacted soils and heavy ground cover (Groninger et al. 2007; Skousen et al. 2009). Further study on options for promoting natural revegetation on reclaimed mine sites is needed. A pre-restoration survey of sites could help identify areas that already have promising natural vegetation development.

Fig. 5 Spontaneous soil formation on overburden under an alder plantation after 30 years (*top*) and spontaneous regrowth on overburden about 25 years after soil heaping (*bottom*); Sokolov Czech Republic; photo: J. Frouz

Facilitating re-establishment of forest understory vegetation

One of the greatest challenges to re-establishment of a natural forest understory community is accessing a diversity of native plant propagules. Concerted research efforts focused on restoration to jarrah forest after bauxite mining in Australia have improved availability of native species seed mixes and understanding of their dormancy mechanisms (Grant and Koch 2007). Unfortunately, in most other regions native plant propagules are not commercially available (Smreciu and Gould 2013) and even if collection is possible, there is often insufficient knowledge of protocols for their nursery propagation (Harrington et al. 1999; Macdonald et al. 2012). Adjacent unmined forest can serve as an important source of plant propagules that can be dispersed to the reclaimed site by wind or animals (Parrotta et al. 1997). However, when large areas are subject to disturbance by mining, there may be no nearby sources of native species propagules for dispersal onto the reclamation site.

Forest floor material (including the litter, fermenting litter and humus layers) and surface mineral horizons) houses a rich bud and seed bank, which serves as the main source of native species propagules for vegetation re-establishment following many natural and human-caused disturbances (Greene et al. 1999; Paré et al. 1993; Schimmel and Granstrom 1996). The potential of this material for use in reclamation was recognized in Australia as early as the 1970's (Grant and Koch 2007) and this approach has gained recent interest in North America (e.g., Mackenzie and Naeth 2007; Cohen-Fernandez and Naeth 2013; Skousen et al. 2011; Macdonald et al. 2015).

Peat-mineral mix (organic layers and surface mineral soil horizons salvaged from nearby peatland forest sites) was initially the most commonly used organic amendment in oil sands mining reclamation (Fung and Macyk 2000). However, use of forest floor material as an organic amendment resulted in better nutrient availability and greater plant richness and abundance than does peat-mineral mix (Mackenzie and Naeth 2010; Mackenzie and Quideau 2012). Forest floor material results in more rapid development towards native upland forest vegetation while peat provides propagules of peatland-associated species (Mackenzie and Naeth 2010; Pinno unpublished). Oil sands sites reclaimed using forest floor material (vs. peat-mineral mix) had higher cover of trees, shrubs, and forbs, and the soil microbial community composition was converging more quickly towards that of natural upland forest (Hahn and Quideau 2013). Plant propagules in forest floor material lose their viability quickly if the material is stockpiled prior to placement but direct placement of forest floor material can overcome this problem and has shown promise in trials in several different locations (Holmes 2001; Iverson and Wali 1982; Koch et al. 1996; Rokich et al. 2000; Tacey and Glossop 1980). For example, direct placement of forest floor material on a coal mine reclamation site in Alberta resulted in rapid ($\leq$3 year) establishment of 65 native species, 30 of which were characteristic of mature, closed boreal forest (Macdonald et al. 2015). For jarrah forests being restored following bauxite mining in Australia, return of about 70 % of native understory species can be achieved through direct placement of surface soil materials (Koch 2007).

Both the depth of salvage and placement of forest floor material influence vegetation re-establishment on the reclamation site. Because seed density declines with depth in the soil profile salvaging to greater depth can dilute the seed bank while salvaging at shallower depths could waste some of the propagule pool (Putwain and Gillham 1990; Tacey and Glossop 1980; Macdonald et al. 2015). Placement depth, in turn, will influence emergence in a species-specific way (Bowen et al. 2005; Holmes 2001; Tacey and Glossop 1980; Zhang et al. 2001). Studies suggest that shallower salvage results in initially higher total cover while deeper salvage can lead to greater richness and favour establishment of native forest species over the slightly longer term (Rokich et al. 2000; Macdonald et al. 2015; Fig. 6). Depending on the amount of available forest floor material, an option would be to place the material at a different depth than the thickness at which it was salvaged. While use of forest floor material for reclamation holds much promise, even when this material rich in plant propagules is used, revegetation success can be poor if site conditions are too severe (Cohen-Fernandez and Naeth 2013).

Seeding understory vegetation

Where native species seed mixes are available, for example for the restoration of jarrah forest on bauxite mining sites in Australia, sowing of diverse mixes of native species can lead to much higher richness and cover of native understory plant species than in unsown area (Norman et al. 2006). Still, recovery towards the pre-mined condition may be very

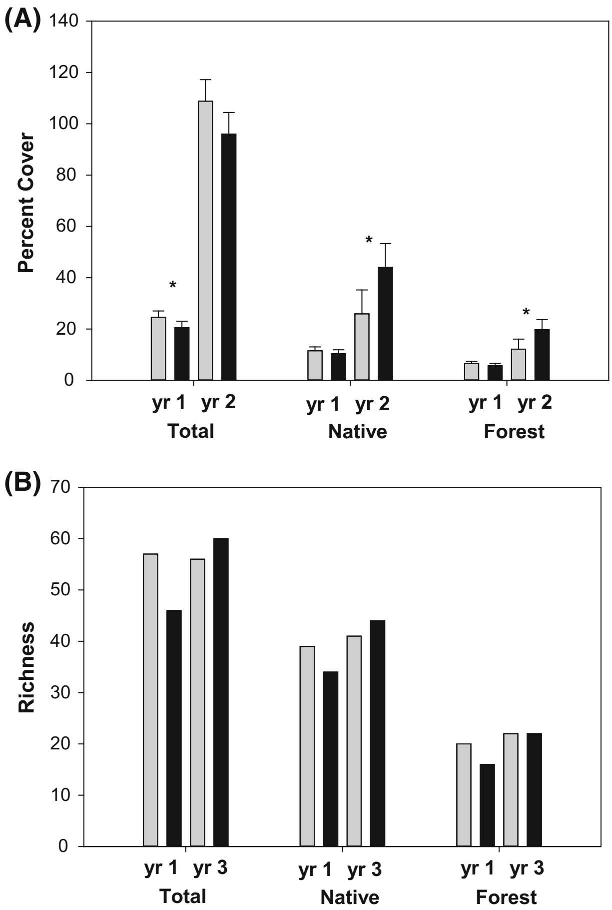

Fig. 6 Comparison of **a** percent cover (mean and standard error) and **b** richness (number of species found in 96 1 m^2 plots) for all species (total), native species and species characteristics of mature boreal forest in year 1, 2 or 3 following direct placement of forest floor material at two depths: shallow (15 cm) and deep (40 cm). *Asterisk* indicates a significant difference between depths for that year. Site was a reclaimed coal mine in Alberta (modified from Macdonald et al. 2015)

slow and the plant community can be strongly influenced by the initial composition (Norman et al. 2006; Koch 2007). During restoration of jarrah forests after bauxite mining in Australia, seeding of native species is often complemented with planting of recalcitrant species to achieve higher total native species richness (Grant and Koch 2007).

The diversity and species composition of a seed mix can be the primary factor influencing the species richness and composition of the resulting community (Grman et al. 2013) and thus requires careful consideration for mined areas slated for reforestation. An herbaceous ground cover is often seeded on reclamation sites to control erosion (Fig. 7). Factors usually considered in the selection of species include time of year, availability and cost of seed, plant functional group, soil and site characteristics, and land use goals (Skousen and Zipper 2010). It is important to establish vegetative cover before the annual period of high intensity storms, when most rill development occurs (Hoomehr et al. 2014)

Fig. 7 Site in Tennessee prepared using the Forestry Reclamation Approach, one year after planting, with coarse woody material visible, and a diverse herbaceous layer that covers approximately 50 % of the ground surface. Photo: J. Franklin

as this can severely hamper vegetative establishment (Moreno-de Las Heras et al. 2008). Therefore the seed mixture should contain at least one species that can germinate and establish quickly to stabilize soils and prevent the seeds of more slowly establishing species from being buried or washed off the site. An annual grain is often used for this purpose, with perennial legumes and grasses included in the seed mix to provide more persistent cover (Skousen and Zipper 2010).

Annual plants provide a cover that establishes quickly which, in addition to stabilizing soil, functions to retain nutrients on the site and provide organic matter, initiating nutrient cycling and soil building processes. Species should be selected based on their ability to germinate and establish in the season in which they are seeded, or as early as possible in the spring, for late seeding dates in northern climates. Native species are typically preferred, but agronomic species are often used for their low cost, reliability over a range of soil types, and well known establishment requirements (Skousen and Venable 2008). Grasses are commonly used as the annual component of the seed mix. Where non-native annuals are used, their persistence or spread can be controlled by using sterile cultivars or species that are not able to complete their life cycle in the climate in which they are planted. Seeding rates should be based on the planting season. Higher seeding rates can be used in later season sowings because annuals will be declining or dead before other species begin to establish. Lower seeding rates should be used when seeding annuals early in the growing season to avoid competition with other components of the developing vegetation.

The objective of establishing perennial vegetation is to provide a ground cover that will eventually be replaced through normal successional processes. Legumes are often included in a seed mixture because of their ability to enrich the soil with nitrogen (e.g., Grant et al.

2007). If native legumes are not available, short-lived agronomic or wild species of low persistence can be selected. Other perennials, generally grasses or native forbs, can be planted to increase the quality of wildlife habitat and diversity of the plant community. Species with taproots are particularly beneficial for loosening compacted soil (Chen and Weil 2010; Skousen and Zipper 2010).

Cover crops that are often used for initial soil stabilization on reclamation sites (OSVRC 1998; Rowland et al. 2009) and can also serve to inhibit establishment of weedy species while providing temporary shelter for natural forest understory species until a sufficient tree canopy develops. Desirable characteristics for cover crop species include the ability to establish quickly, have a diffuse canopy, and nitrogen fixation (Landhäusser et al. 1996). Macdonald et al. (2015) found that *Melilotus officinalis* (L.) Pall. planted as a cover crop suppressed undesirable species while not negatively affecting target forest understory species. However, there was evidence that its biennial nature could result in annual alternation with weedy species, in that case hemp-nettle (*Galeopsis tetrahit* L.).

Several different methods may be used for seeding understory vegetation on a reclamation site, depending on the size of the area and terrain, but choice of seeding method has not been found to influence long-term development of vegetation (Newman and Redente 2001). Hydroseeding is an efficient method for moderately sized sites, and is commonly used on steep slopes or rough terrain where the use of other equipment is difficult (Franklin et al. 2012). Broadcast seeding is generally done by hand and requires no specialized equipment but is obviously suitable only for small areas. Drill seeding uses conventional agricultural equipment to plant seed below the soil surface; this is the preferred method if the soil conditions are suitable but is limited to relatively flat and well-groomed sites. Drill seeding can use lower seeding rates than either hydroseeding or broadcast seeding. Pretreatment of seed to improve germinability can be important for success. For example, it is common to pre-treat seeds of wildfire-adapted species with heat or smoke prior to seeding during jarrah forest restoration following bauxite mining (Koch 2007).

For reforestation, it is preferable to use from seven to 10 species in the ground cover seeding mixture, as this can accelerate the development of vegetation while seeding mixtures of only 3–5 species can delay successional processes (Burger et al. 2009; Kirmer et al. 2012). Often there is a tendency to include too many species in seeding mixtures (up to 25 species!) and this should be avoided because of the waste of seed and the excessive cost. Use of a mixture of genotypes is recommended for the restoration of highly disturbed areas (Lesica and Allendorf 1999) because it can improve the likelihood that some individuals will be tolerant of future conditions. This can be achieved through use of wild-collected seed or mixing seed from several sources or cultivars.

Interactions of understory vegetation with trees

Although herbaceous cover is often needed to control erosion, dense seeded ground covers can hinder tree establishment and growth, slowing successional development (Burger et al. 2005b; Moreno-de Las Heras et al. 2008; Franklin et al. 2012). Herbaceous vegetation competes with tree seedlings for light, water and nutrients. In conditions of low soil nutrient or water availability, root competition has greater influence than above-ground competition (Putz and Canham 1992). Alternatively, the herbaceous layer can facilitate tree establishment by providing sheltered microsites for establishment. The rapid growth rate of many annual and perennial plants also serves to retain nutrients on site and contribute above and below-ground litter. This initiates nutrient cycling and the development of soil biota, and improves infiltration and soil water-holding capacity—all of which aid tree establishment.

The balance of facilitation and competition between trees and ground cover likely depends on environmental factors and the density of the herbaceous cover. Across a range of environments in the eastern USA, survival of tree seedlings planted on mine sites was found to be negatively correlated with the density of ground cover when herbaceous plants covered more than 60 % of the soil (Franklin et al. 2012). At lower densities, the herbaceous layer appears to facilitate tree establishment on some sites while inhibiting trees on other sites. This can be attributed to species-specific differences, neighbor-specific interactions, resource availability and environmental conditions (Eviner and Hawkes 2008). Herbaceous species differ in functional traits and there is evidence that the species composition of planted ground cover influences planted trees. For example, there are numerous reports of non-native forage grasses having a negative influence on tree establishment (Skousen et al. 2009), whereas reports of negative impacts of native species are rare. Plant species composition also influences the soil microbial community on reclaimed sites (Rana et al. 2007), which may have an important effect on tree establishment.

Conclusions and recommendations

Many of the challenges in restoring forests after severe mining disturbances are common to different regions. Research results and practical experience from different regions provide a strong foundation of principles that can be broadly applied to development of best practices for forest restoration. Based on our synthesis of forest restoration approaches we recommend further elaboration of the Forestry Reclamation Approach (Burger et al. 2005a) as follows:

- Laying the foundation:
    - Landform construction should be modelled on natural systems
    - Use and handling of overburden to minimize undesirable effects on soil and vegetation redevelopment
    - Use overburden, surface soils, woody material to create heterogeneity at a variety of scales
- Soil re-establishment
    - Replace topsoil or utilize other organic amendments to more quickly re-establish nutrient cycling and soil development processes
    - Use deeper placement of capping materials on poorer overburden
- Revegetation
    - Leave residual forest patches nearby when possible to serve as seed sources and propagule banks; manage these on an ongoing basis as a source for natural regeneration of forest plant species
    - Facilitate natural regeneration as much as possible

- Align landform, topography, overburden, soil and tree species to create a diversity of target ecosystem types representative of the natural range of variation
- Plan tree planting at the micro- and meso-scale, not at the landscape scale
- Plant mixes of tree species (e.g., early and later successional species) in mosaics at different scales
- Utilize nursery culture and planting techniques to optimize performance of planted trees
- Plant trees as soon as possible on the site and at high densities to help exclude establishment of weedy vegetation
- Utilize direct placement of forest floor material to more rapidly re-establish native forest herbaceous vegetation
- When sowing or planting herbaceous vegetation use native species as much as possible and select mixes that will have minimal competitive impact on trees
- Sow a cover crop of annuals or short-lived perennials chosen for initial erosion control, desirable properties like nitrogen fixation, and growth forms that will exclude undesirable weedy species while not negatively impacting trees or native understory vegetation
- Plan for subsequent entries onto the reclamation site as necessary to facilitate or manipulate stand development trajectories

Success in reclamation or restoration of forest ecosystems post-mining can be assessed by whether the reclaimed forest is productive and self- sustaining and if it fulfills ecological, economic and social objectives. Achievement of this objective relies upon an integrated approach including: landform construction that will support the desired long-

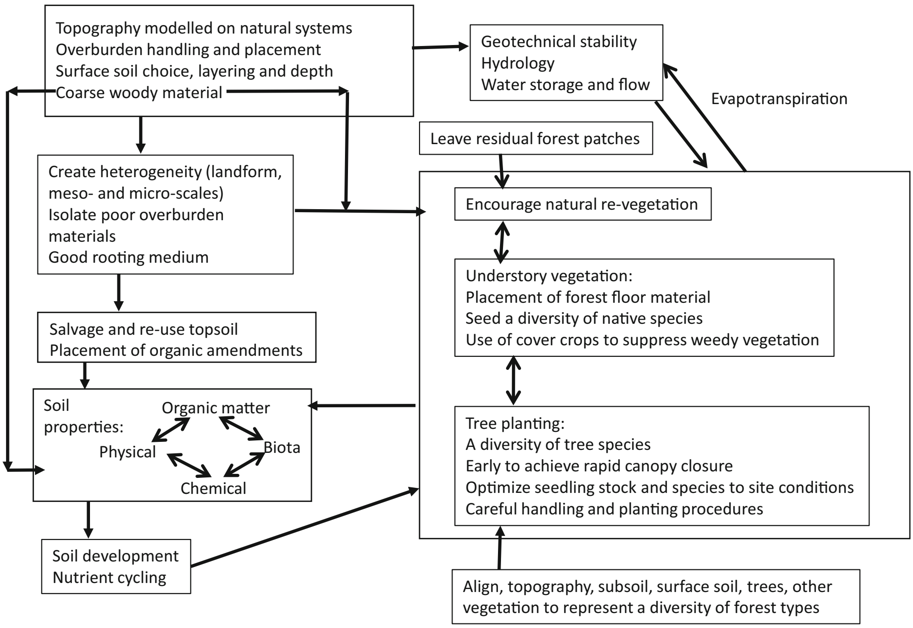

Fig. 8 Conceptual diagram of recommended approaches for forest restoration after severe mining disturbance

term geotechnical stability and hydrologic processes; careful placement and handling of overburden and surface soil materials to facilitate soil development and vegetation establishment; creation of heterogeneity at a variety of scales; matching of topography, overburden, surface soils and vegetation to support development of a diversity of forest ecosystem types; encouragement of natural revegetation combined with use of surface soil materials, seeding and planting of native species (Fig. 8). With an appropriate foundation of soil and herbaceous cover, nearby native species can recolonize the site, and trees will grow rapidly enough to close the canopy and exclude invasive and weedy species. Microbial communities will develop as organic matter pools grow, spurring nutrient cycling. As the plant community becomes increasingly dominated by native species and forest structure develops, enhanced ecosystem services will be provided. Continuous evolution of regulatory frameworks that acknowledge and accept variation in objectives and outcomes will greatly facilitate forest restoration after severe mining disturbance.

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