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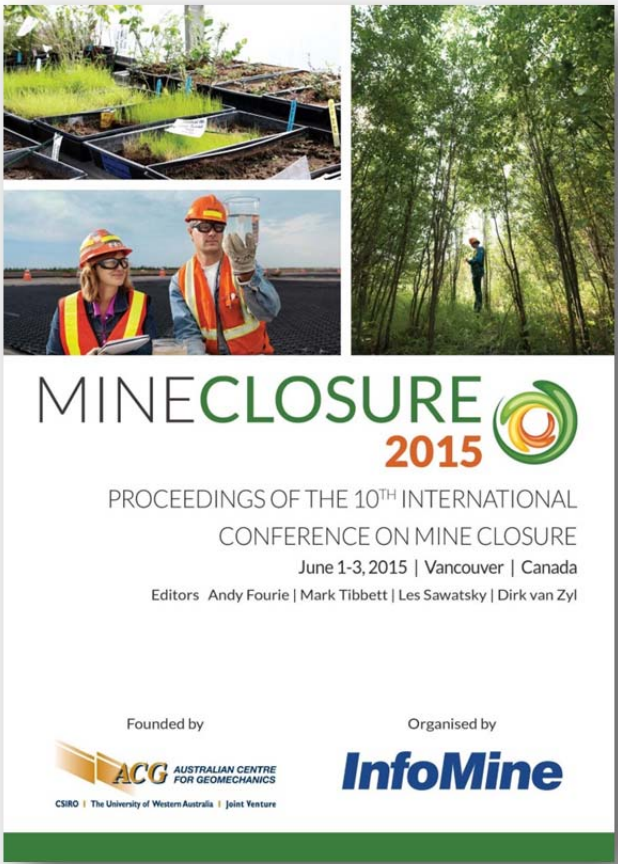

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Does the concept of novel ecosystems have a place in mine closure and rehabilitation?

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Abstract

The validity and practical use of the "novel ecosystem" concept and model, as applied to the restoration and rehabilitation of vegetation and ecosystems, is currently being hotly debated amongst environmental academics and conservationists. The use of the concept and model has been advocated to select alternative target vegetation and ecosystems to those of the pre-mining situations for rehabilitating closed mines in some circumstances.

The purpose of this paper is to examine the concept and its practical use in mine closure rehabilitation planning and evaluation. The underlying basis for the concept and its threshold based model are examined and tested in terms of the arguments for and against it, as proffered by its advocates and critics, using historical and recent evidence from the UK and Australia.

The novel ecosystem concept and its model have merit as a theoretical framework for debate but might not be particularly useful in the mine planning and regulatory context, especially if applied in an uncritical and uninformed manner. For example, there may be risks of excusing poor mine closure outcomes as "novel" when they are in fact wholly inadequate. We give examples of where novel ecosystems seem to work and where the concept seems to fail, and suggest reasons. There is a case for matching more appropriate vegetation and ecosystem targets with some post-mining conditions in the Australian context, and probably elsewhere.

Potential applications of the "novel ecosystem" concept to mine rehabilitation, in the form currently construed, may be a potential risk to sustainable ecosystem outcomes if used as an excuse for restoration failure. Given the lack of clarity and agreement on what constitutes a novel ecosystem, particularly in a post-mining context, we conclude its adoption as a tool for mine rehabilitation is premature and encourage an open debate between regulators, miners, and scientists to provide a safe framework for any future application of the concept.

1 Introduction

The "novel ecosystem" concept and the advocacy of its adoption has rapidly become widely known amongst academics and practitioners in conservation ecology, owing to the publication of numerous articles in the scientific press, as well as a recent book compiling the thinking behind and examples of the concept (Hobbs et al., 2013). The concept has inspired and motivated others, particularly in Australia, to promote its application to mine closure and rehabilitation planning. However, the basis of the concept and its consequences have recently been heavily criticised (Murcia et al., 2014; Aronson et al., 2014). The purpose of this paper is to examine the recently publicised concept of novel ecosystems in the restoration and rehabilitation of vegetation and habitats, and to give a view of the merits of its application in mine closure planning and rehabilitation.

2 Novel ecosystems as a term and concept

The use of "novel" and similar terms in respect of ecosystems is not particularly new. Hobbs et al. (2006) point out that Odum in 1962 had recognised and used the term "synthetic" to describe species assemblages and their ecosystems that had not existed prior to the intervention or influence of humans. Bradshaw et al. (1978) coined the term "novel" in 1976 as a label for deliberately concocted species assemblages (with particular resilience or developmental traits), or unusual establishment techniques or soil amendments that might be adopted to enable the restoration of functioning de novo ecosystems where the soil layer had been entirely lost or severely altered or degraded due to mining activity. Chapin and Starfield (1997) used the term "novel" to describe a native grassland-steppe vegetation that currently exists as a native North American habitat, but which may expand northwards in its geographical distribution and replace tundra vegetation and habitat over the next 150 to 250 years as global air temperatures rise.

Hobbs et al. (2006) and Mascaro et al. (2013) later adopted the terms "novel" and "novelty," respectively, as generic terms to describe species combinations not occurring in a currently recognisable biome that have arisen because of human action. Subsequently the "novel" term was developed into a single unifying concept relating to the reversibility and irreversibility of ecosystem dynamics and transformations (Hobbs et al., 2009). Here, the underlying thinking was that *historical* vegetation and ecosystems had been and were being transformed (due to human influences) to altered assemblages and ecosystems (referred to as *hybrid* types), and in extreme instances to *novel* types where changes in the abiotic environment were so extreme that it was unlikely that either the historical or hybrid types could be reasonably expected to be restored. Hobbs et al. (2009) illustrate this by reference to *historical* Eucalyptus woodlands in southwestern Australia that through livestock ranching had become modified (albeit still having representative main-tree components (i.e. they were a *hybrid* type), and that could revert to the *historical* type through exclusion of grazing and standard conservation measures of introducing lost species by seeding and/or planting. However, when salinization occurred (through a rising water table caused by the deforestation), reversion was unlikely to be achieved by the sort of restoration measures deployed to convert the *hybrid* to the *historical* type in the context of our own life-spans. Under these circumstances, a very different and hence *novel* saline vegetation and functioning ecosystem had developed.

Groffman et al. (2006) have debated the possibility of the occurrence and useful application of ecological thresholds (tipping points) in the restoration and management of vegetation and ecosystems – a mechanism relied upon for novelty by advocates of the current evolution of the novel ecosystem concept. Thresholds are well established and used in the pollution assessment of sulphur, nitrogen, and acid-rain deposition and their potential inducement of biotic changes in sensitive vegetation and ecosystems in the British Isles and Europe (Department of the Environment and Rural Affairs, 2010; Centre for Ecology and Hydrology, 2014). Proponents of "novel ecosystems" frequently refer to notional thresholds and changes in abiotic conditions to differentiate in their conceptual model between *hybrid* and *novel* ecosystem states. The existence and characterisation of the thresholds as a determining mechanism is a point of contention between those who advocate the current concept and those challenging it (Murcia et al., 2014; Hobbs et al., 2014; Aronson et al., 2014). This has led to a perception that it is being claimed that almost all ecosystems have been altered in their composition and are therefore likely to be *novel* or can be treated as being *novel* for restoration planning and conservation evaluation purposes. Accordingly, the notion that most terrestrial ecosystems are currently in a *novel* state (i.e. requiring irreversible change in abiotic conditions) is seemingly implausible as *novel* states (by definition) may only be induced in the most extreme conditions or events such as volcanic eruptions, desertification, glaciation, landslips, or the built environment such as urbanisation, and mining where the soil layer has been lost or severely degraded or changed. Hence, those ecosystems that have undergone or are undergoing biotic change are most likely to be of a *hybrid* type capable of reversion to the *historical* type.

In considering abiotic thresholds inducing *novel* states, and their irreversibility, the important dimensional scale of time in the restoration of vegetation and ecosystems (Humphries, 2013b) is seemingly not factored into the model, even though it is clearly involved and often acknowledged (e.g. Mascaro et al., 2013). It appears that the proponents of novel ecosystems may be applying the abiotic threshold term to a "current" and/or very restricted point(s) in time when actions and interventions might be implemented as a programme of restoration within an individual human's relatively short life-span. The Puerto Rico example cited by Mascaro et al. (2013) is a good example of where the partial reversion to native forests following the abandonment of agricultural land after only 40 years is described as being in a "novel" state owing to its relatively incomplete species composition, whilst acknowledging structurally they "… greatly resemble the historical ecosystem." Over a longer period of time it might reasonably be observed that, as colonisation by the less mobile species takes place, the forest composition would become more similar to the historical. Chapin and Starfield's (1997) predicted changes in tundra vegetation have been cited as an example of a novel ecosystem in support of the current concept and model. However, the change is clearly a gradual process over a considerable period of time (some 150 to 250 years) and this accords with the Gleasonian Framework said by Mascaro et al. (2013) to be a foundation of the novel concept. Similarly, the late E.M. Nicholson (p. 277 in Bradshaw et al., 1978) pointed out that following the catastrophic coastal landslip at Lyme Regis in 1840 it "… looked like a bare quarry for 60 years ..." and by the 1970s it was a mature typical English ash woodland (*sensu* Rodwell, 1991a) and highly valued for its wildlife. Over a longer time perspective, Dimbleby (1978) describes successive European Mesolithic and Bronze Age clearances of forest in England and Scotland, and the subsequent forest regrowth and re-establishment of the original forest composition (from the pollen stratigraphy) even though there was evidence of significant loss and degradation of the soil layer between the periods of afforestation. These examples imply processes of gradients and cascading constraints, seemingly over periods of time, thereby challenging the fundamental notion of the existence of "thresholds" as embodied in the novel ecosystem concept.

3 Application of the novel ecosystem concept to mine rehabilitation

Bradshaw et al. (1978) point out that the restoration of the original type of vegetation and ecosystem functioning in mine closure schemes is unlikely (without intervention) in the short to medium term where there has been a loss of the *in situ soil* resources. Where the soils have been carefully recovered and reused, and appropriate landform and water resources and drainage conditions are reconstructed, similar vegetation and ecosystems are possible. Murcia et al. (2014) in their criticism of the novel ecosystem concept cite the rehabilitation of jarrah forest in Western Australia after bauxite mining as evidence that restoration of severely disturbed ecosystems is possible and the outcome is closer to the *historical* state than being at all *novel*. However, the rehabilitation example cited deploys careful landform reconstruction, soil recovery and replacement, and the introduction of appropriate plant species. In these circumstances, whilst some early differences in composition and ecosystem functioning might be detected, it would be expected according to the Hobbs model that the rehabilitation in the absence of major abiotic change(s) would qualify as being a *hybrid*, *hybrid-historical* or *historical* type according to the degree and importance of the difference, and capability of reversion over time or with intervention.

In contrast, rehabilitated surface coal mines in the Bowen Basin of central Queensland, where soils are often mixed with lower occurring lithologies or are only partially or even not at all recovered, the landform is much altered and the resulting, often sodic or saline, conditions are extreme, resulting in very different composition and stability of vegetation and ecosystem function to the original pre-mining types (Erskine and Fletcher, 2013). In these circumstances, as predicted by Hobbs et al. (2006, 2009), the initial outcome might be considered to be of a *novel* type and of lower intrinsic value relative to the historical landscape and the requirement for landform evolution and its stability (*sensu* Hopper, 2009), but not necessarily so in terms of similar naturally occurring situations or in cases of historic abandoned mining sites where native vegetation and their associated ecosystems are established.

Doley et al. (2012), in setting out to direct the attention of mine operators to more realistic rehabilitation of severely altered landscapes and soils, particularly in arid Australian environments, adopted and developed a version of the "novel-hybrid ecosystem concept." This is subsequently re-rehearsed by Doley and Audet (2013, 2014) with Doley and Audet (2014) advocating the adoption of the concept as a vehicle to drive a change in thinking about mine rehabilitation planning and expectations, in a similar way as Hobbs et al. (2006) were thinking for guiding restoration ecology in general. Doley and colleagues saw the *novel* element of the concept as a means whereby alternative vegetation and ecosystems (that are more appropriate to the post-mining conditions than the *historic* pre-mining types) could be adopted by the industry and regulators to overcome the alleged poor performance of current schemes (Doley et al., 2012). However, in advocating the adoption of *novel* vegetation and ecosystems it is not indicated or specified what the novelty might comprise of in the essential traits of composition, structure, and function in order to guide both practitioners and regulators in the application of the concept in practice – as would be required by mine closure completion criteria in Australia (Glenn et al., 2014). It is in this context that Murcia et al. (2014) and Aronson et al. (2014) contend that the novel ecosystem concept is of no practical use or relevance, and cannot be justified on the basis of current evidence.

4 Testing the concept and model

A test of the value of the novel ecosystem concept and its model is to apply them to actual rehabilitation schemes and to judge the outcome. For example, the Nant Helen surface coal mine began excavation for coal in the late 1980s and part was restored in the mid-1990s (Humphries et al., 1999; Humphries, 2014). The pre-mining *(historic*) vegetation (Table 1) comprised recognisable British native habitat types (Joint Nature Conservation Committee, 1990) and British plant community types (Rodwell, 1991b, 1992).

Table 1 Comparison of pre- and post-mining native vegetation habitats and plant community types after 18 years

	Pre-mining		Post-mining	
Remnant pre-mining vegetation	NCC habitat types*	NVC plant community types#	NCC habitat types*	NVC plant community types#
Semi-improved acid grassland[1]	B1	U4b	B1	U4
Marshy grassland types[2]	B5	M23	B5	M23
Acid dwarf shrub heathland[3]	D1	H12	D1	H12
Wet modified (peat) bog[4]	E1.7	M19 and M25	E1.7	M19a and M25

*Joint Nature Conservation Committee (1990); #Rodwell, 1991b, 1992; [1] Benyon, 2013; [2, 4] Humphries, 2014; [3]Smith, 2012

The excavation for coal has resulted in the almost complete destruction of the acid grassland habitat and communities and the loss of the supporting soil fabric as in the context set out by Bradshaw et al. (1978). Here, rehabilitation in the 1990s was achieved by providing a 0.5 to 1.0 m layer of crushed overburden rock (a Carboniferous sandstone selected as a soil layer substitute) and placed over the Carboniferous mudstone backfill, and an agricultural seed mixture (Humphries et al., 1999). The predicted outcome in applying the novel ecosystem concept and model would be a *novel* type given the significant change in the soil physical and chemical abiotic conditions (Jones, 2013).

Although the original excavation had resulted in extensive loss of the other three vegetation types, recovery of some vegetation and/or soil material was possible. For the marshy grassland, soil with its seed bank was recovered, stored, and reused in the restoration as a topsoil layer over the crushed sandstone soil substitute. The peat fabric and the intact vegetated layer were recovered and replaced in prepared basins (Humphries, 2000). The predicted outcomes for the rehabilitation of these two vegetation and ecosystem types might be a *hybrid* type rather than *novel* given the provision of the supporting abiotic soil component, with the possibility of the *hybrid* types reverting to a *historic* type with or without intervention (Hobbs et al., 2006, 2009). In contrast, the acid dwarf shrub heathland and associated plant species was re-established on crushed sandstone rather than the native soil, and in the context of the concept's abiotic and irreversible threshold, a *novel* outcome would be predicted.

The initial agriculturally sourced and sown prototype acid grassland seed mixture included the productive forage grassland species of *Lolium perenne* and *Poa compressa* as a nurse crop for erosion control measures (Humphries et al., 1999). Almost twenty years on and the sown acid grassland is of recognisable British habitat and community types (Table 1) according to the published criteria (Joint Nature Conservation Committee, 1990; Rodwell, 1992) and comparable to the pre-mining type (Benyon, 2013). In this example, the outcome is not as predicted with the *historic* type becoming re-established rather than the *novel* type or other *novel* type being maintained as a result of the drastic change in soil conditions and loss of plant species. This has come about by the invasion and establishment of native plant species from the surrounding area and propagules brought in by livestock in the grazing of the grassland.

For the re-establishment of the acid dwarf shrub heathland on the overburden material, this involved the introduction and chequerboard replacement of 1.5 × 1.5m turves rescued from the heathland being lost at the time, with reliance on the persistence of the introduced plants to facilitate the colonisation of the remaining bare areas of overburden to recreate the pre-mining heathland habitat and vegetation composition. In 2012, the heathland was surveyed in detail (Smith, 2012) with the outcome that the pre-mining habitat and NVC community H12 type (Rodwell, 1991b) had re-established (Table 1). In this example, the outcome is also not as predicted with the *historic* type becoming re-established rather than the *novel* type or other *novel* type being maintained as a result of the drastic change in soil conditions. This has come about by the invasion and establishment of native plant species from the introduced islands of vegetation.

The marshy grassland and modified bog habitat possibly aligns with the *hybrid* state as a starting point given the recovery and use of the native soil fabric, and it is not surprising that they achieve a *historic* community composition (Humphries, 2014), according to the "novel ecosystem" concept and model (Hobbs et al., 2006, 2009). Similarly, Pywell et al. (2003) have witnessed the same process and outcome for the rehabilitation sowing of native grasslands on natural soils following arable farming in the UK, as have Humphries et al. (2014) in the re-establishment of native floodplain grassland (Rodwell, 1992) on a degraded pasture.

Humphries (2013b) points out that there is necessarily a long timescale of decades and even centuries depending on which vegetation and ecosystem is being considered, before judgement in the case of mine site rehabilitation is possible given the rate of vegetation and structural development, and their life cycles. He suggested that for grasslands this might be in the order of 15 years, for dwarf shrub communities 30 to 50 years and for woodland and forest types 150 to 350 years depending on main-tree composition. That for woody vegetation was illustrated above by E.M. Nicholson's observation (in Bradshaw et al., 1978) about the *Fraxinus excelsior* NVC W8 woodland (Rodwell, 1991a) re-establishment taking in the order of 130 years following the Lyme Regis landslip. The detailed studies on china clay waste (i.e. an absence of in situ native soil cover) in the 1970s and 1980s demonstrated that there is a rapid successional change from an initial *novel* (pioneer) vegetation and ecosystem comprising *Lupinus arboreus* or *Ulex* species – *Sarothamnus scoparius* – *Calluna vulgaris* on the young mineral wastes to recognisable native/naturalised British vegetation of *Betula pendula* – *Quercus robur* – (*Rhododendron ponticum*) NVC W16 woodland (Rodwell, 1991a) on the oldest wastes over a 100 year period (Humphries and Rowell, 1994). Whilst because of the presence of the alien rhododendron, a *hybrid* label might be applied rather than a *historical* type. Putting the presence of rhododendron aside, the succession to a recognisable native type suggests that *novel* types may only be transitory over time on mine sites and drastically disturbed materials, as witnessed for naturally occurring situations, with the time lag depending on their composition and structural traits, as pointed out by Humphries (2013b).

5 Novelty of vegetation and ecosystems for mine closure and rehabilitation

Taking the above findings in the UK example of re-establishing the *historic* vegetation at the Nant Helen Surface Mine Site, it would appear that the novelty and hence *novel* vegetation and ecosystem largely relates, as set out by Bradshaw et al. (1978) and Humphries (1979), to the initial stage of site treatment and revegetation methodology used to achieve a particular target land use and ecosystem.

In the context of Hobbs et al. (2006, 2009), the instigation of *novel* vegetation and ecosystems as long-term outcomes in the UK is probably restricted in scope and detail. This may be related to the much younger and more varied landscape of the British Isles (for example, Rudeforth et al., 1984) than those ancient landscapes and soils in Australia (McKenzie et al., 2004; Hopper, 2009). Novelty and hence *novel* ecosystems may largely be equitable in the UK with alien species and their vegetation assemblages. However, under current European and UK legislation and policy provisions, the introduction of novelty will certainly exclude alien plant species and their assemblages. The importation and establishment of foreign species as trialled in the UK in the 1970s (Jobling and Stevens, 1980) would not now be allowed.

There is a long tradition in the UK of recognising man-made landscapes and soils, and their habitats (Joint Nature Conservation Committee, 1990; Blackstock et al., 2010) and vegetation communities (Rodwell, 2000) as legitimate entities in themselves and ones which often have high nature conservation value, besides the role they have in effective mine and derelict land rehabilitation (Ratcliffe, 1974; Gemmell, 1977; Humphries and Elkington, 1980; Johnson, 1978; Bradshaw and Chadwick, 1980). Mine-like habitat is readily recognised and not usually regarded as a novelty, for example, as "artificial exposures and waste tips," sub-type "spoil" or "quarry" with their mature vegetation described as generic types such as heath, lichen/bryophyte heath, dwarf shrub heath, scrub, scattered trees (Joint Nature Conservation Committee, 1990). The vegetation of open habitats associated with the artificial exposures are also recognised (Rodwell, 2000), for example NVC OV37 *Festuca ovina – Minuartia verna – Cladonia spp* sub-community is associated with heavy metal spoils. Blackstock et al. (2010) recognise slate and other quarry and mining wastes, and their vegetation, as part of the habitat and ecosystem fabric of Wales.

In the UK acceptable long-term *novel* mining related outcomes are likely to be very specific and relate to situations without natural soils or where soil development is expected to be particularly slow, such as rock dumps, toxic spoils, and tailings dams. For example, the stabilisation of toxic mine wastes as spoil heaps and tailings dams with tolerant grass populations (Smith and Bradshaw, 1972; Johnson, 1980; Chadwick and Bradshaw, 1980) has created grasslands of botanical importance alongside their natural counterparts (Rodwell, 2000). Plant communities established for phyto-remediation (Dobson et al., 1997) would be another specific situation. Recently, it has been recognised that coal mine overburden dumps provide valuable habitat for the conservation of endangered lower plant "bio-crust" communities in South Wales (Humphries, 2013a).

The landforms and substrates, and hence the habitats created by mining activities, may be foreign to the Bowen Basin locality, but are unlikely to be unique in the wider Australian geography where it is more likely vegetation and habitat (ecosystem) analogues are to be found. In Australia, broad descriptions as to types are published as the Major Vegetation Groups (MVG) inventory (Department of the Environment and Water Resources, 2007) and more detailed descriptions on a State by State basis (e.g. Regional Ecosystems Descriptions, Queensland Government, 2014). On a national MVG basis, natural types of extreme habitat (similar to those occurring at some mine sites on closure) are characterised as "Other Cover Types," sub-type "Naturally bare – sand, rock, clay-pan, mudflat." At the Queensland State level, comparable natural extreme vegetation and ecosystems appear to go unrecorded.

Given this, in relation to post-mining landscapes, perhaps the novel ecosystem based paradigm referred to by Doley et al. (2012) is not so much about the detailing of the vegetation composition and ecosystem function per se, but alternative habitat and vegetation types being simply "out-of-place" in the Australian context. The "out of place" being the association with other locations such as arid landscapes elsewhere in the Australia or the State, but not locally recognised or documented. It is in this context that there is, for example, scope for exploring novelty (in the current time frame) in the form of adopting as mine closure and rehabilitation outcomes the arid/semi-arid assemblages of plant assemblages which have been successful on mine tailings and rock dumps elsewhere (Tongway and Hindley, 2003; Mulligan et al., 2006). This might be extended to include the recently recognised important dryland biocrusts (Read et al., 2014).

6 Discussion

The "novel ecosystem" concept as advocated by Hobbs et al. (2013) is now hotly debated as to why it is needed and what use it is (Murcia et al., 2014; Hobbs et al., 2014; Aronson et al., 2014). As is only too clear from the above discussion, and as Murcia et al. (2014) and Aronson et al. (2014) point out, there is much confusion in the understanding and application of the "novel ecosystems concept" and its threshold based model. Initially and traditionally, novelty in vegetation and ecosystems was equitable with alien species and vegetation, such as described by Ewel (2013). This is a concept that is easily and universally understood.

Proving the existence of abiotic barrier(s), as a general mechanism impeding the reversion of *novel* to *historic* types, is likely to be central in support of the case for the novel ecosystem concept and the model proposed. The concept will need to factor in time as a key dimension, which may vary from decades to centuries or longer in the case of sodic, saline, metal and other contaminated substrates, and the evolution and eventual stabilisation of appropriate landforms. The Lyme Regis landslip and the china clay sand waste examples, cited earlier, suggest the absence of irreversible thresholds impeding transformation from *novel* to *historic* types over mere decades, making the notion of obligatory abiotic thresholds apparently redundant.

In the test example of re-establishment of the historic acid grassland and acid dwarf shrub heath types, it might be counter argued that the sandstone substitute is locally a precursor of acidic soils, and all that is needed is time for the establishment of plant nutrient accumulation and cycling for these vegetation and ecosystems to become sufficiently established. If this interpretation were to be correct, then the loss of the native soil and substitution of the parent-like geological material would constitute less of an abiotic change than envisaged and more of a biotic change. This suggests the initial rehabilitation state to be of a *hybrid* rather than a *novel* type. Less surprisingly, had the restoration deployed Carboniferous mudstone or limestone material as the soil substitute that occurs within the geological profile and locally, then the outcome would most certainly have been *novel* within the context of the Nant Helen Mine as the resulting mesotrophic and calcareous grasslands, and their shrub counterparts, would have been out of place and not of the *historic* types, although they occur in the wider landscape around the site. In this context, the resulting vegetation and ecosystem types associated with the neutral and calcareous soil materials would not have reverted or would have easily been transformed to the *historic* acidic types without reverting to the measure of soil removal as described by Ewel (2013). This scenario is synonymous with *novel* salinization example cited by Hobbs et al. (2009) and challenges the interpretation by Murcia et al. (2014) of the status of the jarrah forest mine rehabilitation. Perhaps this insight reveals the nature of the so called abiotic barrier referred to by Hobbs and colleagues as necessary for novelty. Clearly, in the mining and soil type contexts, it is not the loss of the soil layer and its substitution with a parent-like material, but the absence of such or its replacement with a material of a very different physical/chemical lithology or origin, which results in novelty. However, from the UK evidence, the resulting vegetation and ecosystems would still be of a recognisable native type and not irreversibly *novel* as a non-native type as required by the Hobbs model, but comprise native and *historical* types that are simply out of place.

Seemingly less debated by the advocates for *novel* states is the outcome of adopting a laissez-faire approach to restoration (Humphries, 1979) or the similar "open-endedness" strategy of leaving revegetation to natural colonisation and successional processes as suggested by Hughes et al. (2011, 2012). This approach to revegetation assumes the tacit acceptance of whatever vegetation and ecosystem outcome occurs in the short and longer terms. In the context of mining, the laissez-faire and open-endedness approaches would embody either leaving the closed mine (following any required initial treatments) to natural colonisation, and vegetation and ecosystem development/successional processes (for example, as described by Rentel and Rentel, 2009). It might be predicted, by default, that the outcome would result in *novel* vegetation and ecosystems of an unusual but native composition (unless it is dominated by alien/introduced species as in Ewel, 2013). However, as Gemmell (1977), Humphries and Rowell (1994), Jones and Jones (2008) and others point out, the novelty of pioneer vegetation establishing on mine and industrial wastes is ultimately likely to be excluded by later invading species and is likely to become a familiar native type.

The outcome of the above is that in the UK context of a long history of naturally revegetated abandoned mine workings, the development of concomitant native vegetation and ecosystems is itself not a novelty. Hence, the use of the "novel ecosystems" concept and its threshold model are likely to be of limited relevance and application. In undertaking the rehabilitation of mine sites in the UK, the general use of the term novel in the context of Bradshaw et al. (1978) is applicable and of practical use without the need to refer to that of Hobbs et al. (2013). Elsewhere, like Australia, where there is a shorter history of naturally vegetated mine workings and/or less interest in the recording of the resulting vegetation and their ecosystems, *novel* may be a better description of the state of human knowledge rather than novelty per se.

Given the historical context, the rehabilitation of specific vegetation and ecosystems types associated with the legacy of mining activities is recognised in the UK and may be acceptable (with appropriate evidence) to the planning and regulatory authorities, and can be technically incorporated into mining consents. The same is apparently the case in Australia. Here, government guidance to the mining industry (Department of Industry, Tourism and Resources, 2006) readily recognises that mining operations can result in a range of features, such as waste rock dumps, tailings storage facilities, water filled voids etc., and that rehabilitation and land use targets should take into account the specific characteristics of the climate, landform, substrate etc. The guidance advocates the zoning of the features as separate entities ("domains") and the setting of specific rehabilitation goals and targets during the mine closure planning stage. It is accepted that these may be different from the pre-mining land use and vegetation. This includes the use of pioneer vegetation and ecosystems and reliance on successional processes for the targeted outcome. The provision for non-pre-mining land use and vegetation is translated into the provisions of current state government guidance (e.g. Department of Mines and Petroleum and Environment Protection Authority, 2011; Department of the Environment and Heritage Protection, 2014). It is within this context of aligning rehabilitation to the physical and chemical requirements of the target vegetation and ecosystem types that Doley and colleagues argue their case for adopting (*novel*) alternatives for mine closure and rehabilitation for a more successful outcome (where vegetation and ecosystems are better matched with site, landform and substrate conditions). Whilst the principles are well understood and provided for in the Australian planning and permitting process, what is seemingly missing is the identification of the viable alternative vegetation and ecosystems that provide the requisite stable and self-sustaining conditions consistent with the nominated post-mining land use (Department of Industry, Tourism and Resources, 2006). To this end, there appears to be a gap in the knowledge that needs a systematic research programme to define the target appropriate habitat, vegetation composition, and ecosystem types for the domain types associated with mining operations in Australian, and possibly elsewhere too. For example, bio-crusts might be an alternative and novel kind of "vegetating" mine tailings, instead of the conventional practice of seeding or planting local forest types. In order to address the concern that the adoption of the novelty is one of faith and open to criticism of "lowering the bar" for rehabilitation standards (Perring et al., 2013; Perring et al., 2014), there will be a need for long-term monitoring programmes of pilot and eventually established schemes in order to create the kind and level of understanding that is well established for former mine sites and wastes in the UK.

7 Conclusions

Accepting that mine workings can be "out of place" in the landscape, as described by Erskine and Fletcher (2013), the establishment of non-locally occurring vegetation and ecosystems characteristic of the pre-mining landform and soils would be a local novelty and might be *novel* in this context. So, in response to the question posed by this paper: "is there a place for the 'novel ecosystem' concept in mine closure and rehabilitation?" The answer is "yes" in principle as there is provision in the current best practice and regulatory guidance, but in practice it is a "no" in the context it is currently being marketed, given the confusion around the concept and model and apparent lack of supporting evidence (Aronson et al., 2014). It might be misleading and unsafe in mine planning and regulation terms for the present time for it to be adopted without further critical scrutiny.

Could it be useful as a debating tool in mine rehabilitation planning? The answer is "yes" as Doley and Audet (2014) suggest there is benefit for a more active airing as to what is to being proposed for mine rehabilitation rather than to adopt the pre-mining historical vegetation by default. Do we currently know enough about the type and composition for, and the sustainability of, rehabilitation alternatives for mine sites? The answer is probably “yes” in the UK, but possibly “no” in the Australian context and maybe elsewhere too, where alternatives to the pre-mining situation might be viewed as a novelty.

Given the above, is there a way forward? “Yes” there is. If novel ecosystems are to be considered in mine site reclamation, some fundamental rules need to be established. Firstly, Mascaro et al. (2013) set out clear direction that ecosystems are not ones that are part of: “… a historical range or variability … intensively managed systems … managed with the purpose of reproducing historical ecosystem …”. Secondly, reconstructed novel ecosystems should be expected to have ecosystem services greater than or equal to the historic system and its own intrinsic demonstrable ecological value unless, thirdly, alternative candidate systems are adopted from ecosystems with analogous landforms and soils, and possibly also from abandoned historic mine sites. In these cases, in areas of high conservation value, local stakeholders and regulators may not accept these novel systems, but see them as alien to the region.

As a final word on the “novel ecosystem” concept and its application to mine rehabilitation, in the form currently construed, it may be a potential risk to sustainable ecosystem outcomes if used as an excuse for restoration failure. Given the lack of clarity and agreement on what constitutes a novel ecosystem, we conclude its adoption as a tool for mine rehabilitation should either be avoided or used with caution for the present time. However, the usage of the term to describe techniques and methodologies in the context of Bradshaw et al. (1978) is simple to understand and has practical application.

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