



The creation of compensatory habitat—Can it secure sustainable development?

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Summary

We review the effectiveness of habitat creation in an attempt to better understand the potential for delivery of compensatory habitat as part of sustainable development solutions, or for the establishment of habitat banking. Our review highlights considerable differences in the timescales needed to create conservation habitat of a comparable quality. Some wetlands may take just a few years, some grasslands of nature conservation value are known to be relatively young (<80 years old), but woodlands may need to be hundreds of years old before they achieve a similar level of interest. Our knowledge of the abiotic requirements for some habitats, for example hydrological conditions for alkaline fens, is poor and suitable conditions are rare, making re-creation of such habitats highly problematic.

Some faunas, such as some dragonfly and water beetle assemblages, may be readily catered for; others are dependant both on structural aspects of the habitat and on the mobility of individual species, and are far more difficult to accommodate, e.g., invertebrates associated with ancient trees. Historic examples of habitat regeneration are poor models of habitat regeneration on modern arable soils. Considerable changes in soil structure, pH and chemistry have resulted from the introduction of modern soil preparation techniques, fertilisers and pest control. Recent studies also suggest that mycorrhiza are fundamental to establishment of many habitats of conservation interest.

Compensatory habitat creation can probably be used in some wetlands and inter-tidal environments, but the prospects for success in many terrestrial situations are far less certain. It therefore follows that compensatory habitat creation (also called “offsets”) cannot be relied upon in all circumstances as means of offsetting loss of the highest quality habitat, and cannot be seen as a consistent and reliable delivery mechanism for sustainable development.

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Introduction

Habitat creation is widely adopted as a measure to offset losses, either as a result of infrastructure and commercial development pressures, or in response to declines in the status, extent or distribution of particular biotopes, species or assemblages. A general review of UK experience is provided by [Parker \(1995\)](#). In this review we concentrate on the issues as they pertain to the provision of compensatory habitat where developments damage or destroy primary nature conservation interest and compensatory measures are undertaken to offset that impact of habitat loss in order to achieve sustainable development ([World Commission on Environment and Development, 1987](#)). The range of habitats covered by this review is by necessity a sub-sample of those that might be encountered during evaluation of development proposals and has been chosen to illustrate particular points rather than to provide strict guidance on what is or is not possible.

Habitat creation (in locations where there is no historic record of its occurrence) or re-creation (on sites where similar habitat formerly occurred) have been used in conjunction with habitat protection legislation in the UK and elsewhere to underpin remaining populations or habitats. An extension of this approach is the concept of habitat banking or mitigation banking, which is an established mechanism for offsetting commercial development of wetlands in North America where new wetlands are created or 'restored' and sold as credits to offset loss of other wetlands by other developers at some later stage ([Crooks & Ledoux, 1999](#)).

Habitat banking also has its proponents as a possible mechanism for delivering sustainable development in the UK and Europe. In the European examples the conceptual proposal is a variation on the North American model; it involves the creation of new habitat by a particular developer at a specific location and drawn upon by that same developer to offset losses due to its own development projects. There are, however, no current examples of habitat banks in the UK because it is a conceptual idea with no legal foundation for implementation.

Our particular interest in compensatory habitat creation was stimulated by ongoing casework arising from new port development but it has wider implications. This is because a wide variety of habitats come under pressure from development and what may be practical and acceptable in one scenario may not be acceptable in another. In particular, we must ask whether replacement habitat will be of similar quality to that lost to

development or of sufficient quality to support vulnerable plants and animals that are the subject of restoration initiatives.

Some elements of the natural environment can clearly be restored, created or re-created while there are others for which there is limited evidence of re-creatability. Amongst the creatable elements are numerous water bodies of recent origin, ranging from extensive lakes and flashes arising from mineral extraction and mining subsidence, to small ponds dug to enhance the landscape or for watering livestock. A great many of these are now designated in Great Britain as Sites of Special Scientific Interest (SSSI) either for aquatic plant and animal assemblages, as waterfowl breeding and feeding sites, or for particular rare species such as *Triturus cristatus* (Laurenti) or *Botaurus stellaris* (Linnaeus). Some exceptional examples include the extensive reedbeds of Stodmarsh National Nature Reserve (Kent), Far Ings National Nature Reserve (North Lincolnshire) and Shibden Pond SSSI (near Newcastle).

The English coastline provides further pointers to the re-creatability of some habitats. Coastal salt-marshes have evolved on sites in Essex and Suffolk following breaches of sea walls and flooding of agricultural land ([Morris, Reach, Duffy, Collins, & Leafe, 2004](#); [Wolters, Garbutt, & Bakker, 2005](#)). Behind modern sea walls, there are many kilometres of broad, shallow borrow dykes from which fill was extracted to construct sea defences. Today, many of these support rich assemblages of brackish-water plants and animals, and are also known to support many of the suite of species commonly associated with saline lagoons ([Essex Field Club, 2005](#); [Lincolnshire LBAP, 2005](#)).

Throughout Great Britain it is possible to find some species-rich semi-natural grasslands that exhibit distinct ridge and furrow features typical of the communal agriculture of the Middle Ages. Areas that were once arable have developed into species-rich grassland over several centuries. In addition, under some specific conditions, botanically diverse grasslands have developed over shorter timescales of several decades ([Gibson, 1998](#); [Gibson & Brown, 1991](#); [Wells, Sheail, Ball, & Ward, 1976](#)).

The creation of some types of site of value for nature conservation is therefore possible. Equally, there are obvious examples of habitat that develop over geological timescales. These include limestone pavement, raised mires that have evolved over thousands of years, and base-rich flushes that depend upon specific, localised hydrology and lithology. Such habitats reflect a series of very specific conditions that have evolved over the

period since the last glaciation and they do not figure further in the debate because their key features are not re-creatable in realistic time-scales.

Current experience: selected examples of habitat creation

The construction of a pond offers a simple insight into the speed with which some plants and animals will colonise naturally transitory habitats. For example, a pond created in 1985 by removing over-lying rubble from former river alluvium at Bennetts Hole Local Nature Reserve in South London was colonised in the same year by *Nuphar lutea* (Linnaeus) and various *Callitriche* spp. (R.K.A. Morris, unpublished observation). These species almost certainly arose from germination of a relict seed bank of some considerable antiquity (50 years or more). Even a reputedly scarce species such as *T. cristatus* has colonised newly created habitat, as it has colonised many recent shingle workings at Dungeness, Kent, to such a degree that an internationally important population is present. Mobile species such as dragonflies, some aquatic Hemiptera and many water beetles will also rapidly colonise (Moore, 2002). Much more remarkably, a wide range of species, normally associated with brackish conditions that might be expected in coastal localities, have been found to colonise inland sites such as Mickletown Ings SSSI whose water arises from saline run-off from the adjacent colliery tip (Lunn, 2001). However, early colonisation does not necessarily mean that the assemblage of plants and animals continues in its original form. This is amply demonstrated by changes in dragonfly faunas over lengthy timeframes (Moore, 1991, 2001).

Development of vegetation on former mineral workings such as gravel pits, quarries and spoil heaps of various kinds, provides a further clue about the ability of plants and animals to colonise "unoccupied" environments. Barnack Hills and Holes National Nature Reserve, is limestone grassland that has developed on the over-burden left after limestone extraction ceased in the Middle Ages. Other important localities of more recent origin include Herald Way Marsh near Coventry, a former pulverised fuel ash (PFA) from coal-fired power station tip that supports a rich invertebrate fauna of conservation importance (Falk, 1998).

In the UK, there has been much interest in the creation of saltmarshes, mudflats and freshwater reedbeds. In the case of saltmarshes, there is

robust evidence to support the contention that new habitat can be created and that this will happen relatively quickly (Atkinson, Crooks, Grant, & Rehfish, 2001). Examples of successful saltmarsh creation include managed realignment at Freiston Shore (South Lincolnshire) and at Tollesbury (Essex) (Morris et al., 2004). Creation of reedbed habitat in former mineral workings is extensively demonstrated at a variety of locations: for example at Barton and Barrow claypits (North Lincolnshire), Ham Wall Nature Reserve (Somerset) on former peat workings, and at Dungeness (Kent) in former shingle winnings.

Grassland creation has been the subject of considerable interest since the early 1970s and currently there are targets for re-establishing different types of semi-natural grassland in the UK Government's Biodiversity Action Plan (UK Biodiversity Group, 1998). Floristically rich grasslands resembling ancient semi-natural reference communities have developed on former arable land in the past (Wells et al., 1976); although Gibson and Brown (1991) and Gibson (1998) have shown that neutral and calcareous grasslands that closely resemble ancient semi-natural grassland take a minimum of 100 years to develop. However, the conditions under which such grasslands formed were very different to those pertaining today. As a consequence of this, most projects to restore botanically diverse grasslands involve the introduction of seed or plant material, usually coupled with modification of the soil environment to reduce nutrient levels. Using such techniques it has proved possible to create facsimiles of semi-natural grassland in a matter of decades. However, while such grasslands may superficially resemble their ancient semi-natural counterparts, subtle differences in species composition and vegetation pattern remain. In addition, the re-assembly of invertebrate assemblages is a much slower process (Mortimer, Booth, Harris, & Brown, 2002) and we know little about the development of other ecosystem components and processes such as trophic interactions, soil microbial and fungal communities and nutrient dynamics (Walker et al., 2004b).

Heathlands have been the subject of extensive re-creation efforts, with the UK Biodiversity Action Plan setting a target of 6000 ha of new heathland creation. In Denmark (Degn, 2001) an arable field left to revert over 22 years is now recognisable as heathland. In the UK, however, results have been mixed, depending on the original situation (Pywell, Wadsworth, Cooper, & Smith, 2002; Walker et al., 2004a). Different degrees of failure are reported for projects at Minsmere (Suffolk) (Mitchell & Hare, 1999; Owen & Marrs, 2000), at Dunwich (Suffolk)

(Dunsford, Free, & Davy, 1998) and in Dorset (Pywell, Webb, & Putwain, 1995; Smith, Webb, & Clarke, 1991). This is not to say that the "final" habitat has no interest; only that valuable lowland heathland may not be easy to re-create from arable land. The problems originated from the persistent effects of past liming, which kept a high pH; and the high cost and short-term effects of acidification, which resulted in vegetation types which differed from the target heathland. High concentrations of ruderal species and those adapted to disturbance and/or high nutrient levels in ex-arable land are reported by Pywell et al. (1995) and Dunsford et al. (1998). On the other hand, heathland creation on former mineral workings in Cornwall and Dorset (Jenkin, 2000; Larson, 1999; Pywell, Putwain, & Webb, 1996) gives rise for concern. Other promising examples of heathland creation have involved felling conifer plantations and stripping off humus layers (Pywell et al., 2002). However, longer-term monitoring is needed before a positive outcome can be confirmed.

Deliberate woodland creation has been carried out for centuries. Mature stands of native trees and shrubs, both those planted on open ground and naturally colonised areas, can develop rich assemblages of plants and animals. More deliberate efforts to mimic semi-natural communities have involved matching of mixtures to soil and lithographic variation, varying planting patterns, transplanting seedlings or even whole coppice stools and moving samples of woodland soil (Buckley, 1989; Ferris-Kaan, 1995; Helliwell, 1996; Rodwell & Patterson, 1994). The most valuable wildlife sites tend however, to be longest established: even more so than grassland and heathland sites. This is because of the time taken for the trees and shrubs to go through their life-cycle (anything from 50 to 500 years), the slower development of woodland soils, and the apparently poor dispersal capabilities of many characteristic plant and animal species. A further conservation factor, though only in part an ecological one, is that the distinctive features of individual ancient woods may include or be the result of centuries of management. This history cannot be replicated: the cultural and physical environment in which new woods develop will inevitably be different. Furthermore, the climatic condition under which old forests developed may have been quite different from present conditions. Therefore ancient woods are not re-creatable: our objective can only be to create a wood that may eventually have a similar level of value. Yet, because of the time-scales involved we can never know if the process will be successful.

Where should compensatory habitat be created?

Examples cited so far involve habitat creation largely as a by-product of changes in land management or other exploitative activities, where important habitat creation has been an incidental result. At a practical level, the development of compensation packages for flood defence works and major ports under the Habitats Directive (EEC, 1992) has helped to identify some of the issues that should be taken into consideration when habitat creation is needed to deliver specific objectives. To date, such compensation packages have largely focused on the provision of new mudflat for feeding waterfowl and therefore concentrate on the ability of new habitat to maintain numbers of birds.

The most advanced port compensation package is at Trimley in Suffolk, which was delivered as compensation for channel deepening in the approaches to the Stour and Orwell Estuaries by Harwich Haven Authority. More recently, the Environment Agency's realignment at Paull Holme Strays on the Humber Estuary (North East Yorkshire) has been surprisingly successful in its first year (Richardson, 2005). There are continental examples of broader-scale compensatory habitat creation such as works to offset expansion by the port of Bremerhaven. These particular cases seem to have been successful, but Atkinson et al. (2001) caution that some examples of inter-tidal habitat creation have taken longer than expected, and that there can be no certainty that offsetting habitat creation will be successful in all cases. Each case has been different and there is no simple solution to defining the scale of necessary compensation, but there are some broad principles.

Compensation should, wherever possible, be adjacent to the existing site so that detrimental impacts are properly offset and the designated site is returned to favourable condition. In the event that there is no scope for such measures adjacent to the affected designated site, a wider area of search has to be considered, with the objective of securing compensation adjacent to another designated site within the same biogeographic zone. As the area of search widens, the implications of habitat loss on the affected site increase. If suitable compensatory measures cannot be provided in a location contiguous with the site that is affected, the replacement habitat is designed to ensure the overall coherence of the Natura 2000 series. In such circumstances the affected site cannot be returned to favourable condition—any loss may be permanent if compensation is not contiguous with the damaged site. As a

consequence the scale of compensatory measures is should be increased to give some assurance that the actual numbers of organisms supported is commensurate with the lost site.

While it may be desirable to create compensatory habitat in advance of damaging activities, as in the European variant of the model for habitat banking, it is not always possible in practice. In such circumstances the equation considering losses and offsets must take account of the detrimental impacts of delays in securing wildlife interest of a similar value. Part of this process involves the provision of greater areas of habitat of lesser quality, although this habitat may be expected to improve over time. There are, however, no established fixed ratios for compensation and packages and existing compensation packages have been determined on the basis of local judgement, the nature of the impact, temporal issues and the nature of the receptor site.

Possible factors influencing successful terrestrial habitat re-creation

Agricultural intensification and land drainage have been accompanied by declining areas of grassland and some wetlands. For example, studies of Thames Estuary grazing marshes have shown that the extent of such marsh has declined from 13,300 to 4600 ha (65%) between 1935 and 1989 (Thornton & Kite, 1990). Similarly, it has been estimated that 97% of lowland unimproved grassland was lost between 1930 and 1984 in England and Wales (Fuller, 1987). Heathlands too have suffered considerable loss; in Suffolk 90% was lost between 1783 and 1983, and in Dorset of 87% was lost between 1759 and 1987 (Farrell, 1993). Ongoing loss and fragmentation of remaining semi-natural habitats may affect restoration potential, and hence the success of habitat creation projects. At a landscape scale, the opportunities for natural passage of plants and animals are more restricted than previously (e.g. Bourn, Thomas, Stewart, & Clarke, 2002). This is particularly important when using examples of habitats at sites that have reverted from a previous agricultural or industrial use, when the surrounding landscape was less fragmented, as a guide to what might be possible in terms of modern replacement habitat.

The abiotic environment should be matched as closely as possible to the desired outcomes. The relationship between plant distribution and drift and hard geology and of hydrology is well established, as illustrated by Hopkins (2003) and Louse-

ley (1976). Similarly, some animals are clearly associated with particular geology (e.g. Morris, 1998) and vegetation structure (Gardner, Hartley, Davies, & Palmer, 1997; House & Spellerberg, 1983; Webb, 1989). These factors, together with others such as aspect (e.g. Key, 2000) and regional variation in rainfall and temperatures, mean that a wide variety of parameters must be taken into consideration when designing a habitat creation project as a response to possible loss of conservation quality habitat.

Some elements of the new habitat may be deliberately introduced by planting, for example field layer species in some woodlands (Francis & Morton, 2001). This raises philosophical questions (e.g. Townsend, 2005) as to whether such introductions detract from the "naturalness" of the outcome. Deliberate introductions remove the competition that occurs at the seed germination and seedling stage and artificially constrains at least the initial distribution of individuals and species. There are also practical questions of how successful such introductions are, and the costs involved. However the majority of the animal assemblage will have to arrive by natural colonisation. Therefore, the location of the "target" site must be seen in relation to existing habitat taking into account the mobility of key species and any possible reliance on continued interchange between habitat patches (e.g. as meta-populations).

The importance of plant mobility (or apparent lack of mobility) has also been highlighted through the use of indicator species to evaluate woodland origin (e.g. Peterken & Game, 1984). Gibson and Brown (1991) showed that even in close proximity to existing chalk grassland, and provided management and soil conditions are appropriate, it may take many decades for a similar assemblage of species to become established. *Silene silaus* (Linnaeus), for example, rarely disperses seeds beyond a few metres from the parent plant (Bishoff, 2002). Therefore, when grassland creation is undertaken at a distance from a source pool of typical species, the chances of creating comparable habitats to those on ancient sites are significantly reduced, making this a key factor in the species composition of regeneration (Walker et al., 2004b).

Few studies have looked at invertebrate colonisation of grasslands created using different seed mixtures on ex arable land in experimental plots. Those that have been conducted are largely confined to above-ground species and not to soil dwellers (e.g. Mortimer et al., 2002; Christal, Davis, Earren, & Macintosh, 2001). Invertebrate assemblages associated with newly created grassland appear to be distinct from those characteristic

of semi-natural grassland communities. As might be expected, early colonists of new grasslands are mobile generalists typical of widespread agricultural habitats.

The prospects for re-colonisation from distant colonies must be much lower for weakly mobile or brachypterous species such as the fly *Stilpon graminum* Fallén (Dipt. Hybotidae) or the moth *Lycia zonaria* (Denis & Schiffermüller). This may also apply to specialist parasitic species within fly genera such as the Bombylidae (bee flies) and Tachinidae, and within the parasitic Hymenoptera such as the Chalcididae. Importantly, many of these species belong to relatively poorly known families with limited data on host associates or life histories.

The factors behind the high differentiation between ancient and relatively young woodland beetle faunas are complex, but Speight (1989) emphasises the role played by habitat fragmentation in restricting dispersal or re-colonisation by specialist saproxylic species whose mobility is limited. Few studies of colonisation of newly created woodland have compared these stands with young growth in existing woodland: but clearly there is little point in expecting new woods to support invertebrates associated with old trees. Even in the case of species associated with recently cleared coppice, many such as *Mellicta athalia* (Rottemburg) are notoriously poor colonisers that require a continual supply of newly cleared habitat within close proximity of each other (Warren, 1991).

Past examples of successful colonisation may not reflect future prospects. The majority of conservation quality grasslands in the UK arise from the period prior to 1939 and the introduction of large-scale mechanisation and artificial fertiliser and pesticide application. In addition, soil fertility has been shown to have increased from atmospheric sources (Bobbink, Hornung, & Roelofs, 1998; Yesmin, Gammack, Sanger, & Cresser, 1995). Thus, projects now seeking to create new grassland and heathland have to contend with greatly changed soil chemistry, and soil structure problems exacerbated by our modern ability to deep plough, as well as a more fragmented and species-poor countryside.

Our understanding of the significance of soil microbial assemblages in restoration ecology is incomplete. There is, however, increasing evidence that the nature of soil microbial communities may influence the establishment and persistence of some plant species (Gange, Brown, & Sinclair, 1993; Michelsen et al., 1999; Yesmin, Gammack, & Cresser, 1996). There has long been recognition

that ensuring mycorrhizal infection is valuable for establishment of trees on long-unwooded sites and Merryweather (2001) suggests other roles for soil micro-fungi in new woodland. Soils previously in intensive agricultural use have reduced biomass, richness of mycorrhizal fungi and of bacteria brought about by past use of artificial fertilisers (Van der Heijden et al., 1998; Walker et al., 2004a, b). These differences from traditional low intensity agriculture as practised prior to the Second World War must be recognised as potentially serious impediments to habitat creation and colonisation by natural means.

Evaluation of possible success criteria

When compensatory habitat creation is proposed, there must be certainty that specific biological parameters can be met before there can be confidence that lost communities will have been adequately replaced upon completion. These requirements need to be seen in the context of achieving sustainable development solutions for development projects, rather than the broader context of securing improved biodiversity opportunities where tailored habitat creation or restoration for a single species, or perhaps a suite of species, is the objective (see Bakker, Grootjans, Hermy, & Poschlod, 2000).

In many cases of compensatory habitat creation the principal objectives of a project will focus upon plant communities. Under such circumstances the replacement site must be similar in overall species richness, assemblage composition and distribution of rare or more specialist species within the site. The time-frame needed to deliver this criterion may take many years and some plant species such as those with stress tolerance or with affinities to infertile habitats tend to perform badly on restoration sites (Pywell et al., 2003). Intermediate milestones are needed to ensure that the successional processes are on track, especially as early indications of success may not be borne out by longer-term monitoring (Edgar, Griffiths, & Foster, 2005; Gibson, 1995).

Where important invertebrate assemblages have been recognised at the site that is under threat, the replacement site must be shown to support similar assemblages, typical of that particular habitat and supporting the rare or scarce elements of such assemblages. This is more likely to be successful in the case of strongly thermophilic assemblages associated with ruderal communities, but is much more problematic in the case of communities that reflect long-term habitat

continuity. In the majority of cases, environmental impact assessments devote relatively little effort to invertebrate communities and other “minor” taxa such as lower plants, and as a consequence the baseline understanding of the assemblage to be replicated is usually insufficient to make a realistic judgement of success.

Success criteria for single-species packages, such as self-sustaining populations of *Triturus cristatus* (Linnaeus), some breeding birds and mammals, and as feeding grounds for over-wintering waterfowl, are potentially more straightforward because the measures of success can be readily quantified and reported. This is especially true if there are baselines of numbers displaced that can be correlated with survey data for the compensation site over a particular timeframe. Even with carefully defined success criteria post-project monitoring and reporting is not always completed, as demonstrated by Edgar et al. (2005). Their paper gives cause for concern because it is difficult to determine the degree to which offsetting and monitoring measures have been implemented. At a time when reliance on translocation and habitat creation is increasing, such uncertainties and failures make it difficult to maintain confidence that offsetting measures set as a condition of a particular development consent are being delivered to the necessary standard.

Success criteria should take account of possible shifts in the distribution of some species. Existing and newly created habitat will inevitably be affected by shifting patterns of distribution and of the degree to which target species and assemblages are affected by such factors as short- and long-term climate change. For example, periods of prolonged drought can lead to changing vegetation structure with drought tolerant species such as *Rumex acetosella* (Linnaeus) and *Plantago coronopus* (Linnaeus) gaining temporary dominance on some Thames basin acid grasslands (R.K.A. Morris, unpublished observations). The distribution of many thermophilic invertebrates in Great Britain (and northern Europe) is clearly changing, with species such as the wasp *Philanthus triangulum* (Fabricius) (Hym., Sphecidae) and the hoverfly *Volucella inanis* (Linnaeus) (Dipt., Syrphidae) clearly expanding their ranges (Edwards, 1997; Morris & Ball, 2004). Detectable change is perhaps most apparent in a range of larger, conspicuous and more mobile invertebrates, but must be expected to follow similar paths across a wide range of organisms.

Existing locations for some species can be expected to become less viable. Suitable conditions for *Fagus sylvatica* (Linnaeus) may decline in south-

east England in the next Century (Berry et al., 2001) because of climate change. Is it therefore appropriate to promote creation of new beech woodland in south east England, either as part of the Habitat Action Plan expansion targets or as compensation for loss of existing habitats? Should an alternative habitat be promoted for that region, or should the “beech” woodland creation be promoted in another part of the country where the climate may remain suitable (Wesche, 2003)? These few examples give an indication that the plants and animals that compensatory habitat creation seeks to accommodate may cease to be viable in one area and might be better accommodated elsewhere according to the implications of climate change.

Conclusions

Habitat re-creation does have the potential to contribute to positive action for stabilising and reversing some biodiversity loss, particularly when carried out in the proximity of existing habitats. Ruderal and early seral stages are more likely to be replaceable than longer-established and ancient habitats, especially those that are reliant upon specific management intervention or development of ancient micro-habitat such as mycorrhiza-rich soils, subterranean rotting timber or rot holes. These older and more biologically specialised resources are more difficult to replicate or replace. Furthermore, there are species whose habitat is likely to prove almost impossible to re-create and compensatory measures are unlikely ever to succeed. These conclusions seem to be consistent with those of Harris and van Diggelen (2006, pp. 3–19) who argue that “simple recreation of past species lists is unlikely to succeed” and go on to identify some clearly achievable goals such as “reinstatement of a hydrological regime or re-introduction of a keystone species” as more realistic goals.

These are important considerations in the development and application of planning frameworks that affect nature conservation sites. The tests of Article 6(3) of the EC Habitats Directive (EEC, 1992) are designed to reflect these issues, but do not specifically test the degree to which lost habitat can be re-created. The re-creation potential of particular habitats therefore needs to figure strongly in the decision-making process.

Habitat creation seems to offer a suitable tool in the case of a narrow suite of habitats and species for delivery of offsets for habitat loss in order to

achieve sustainable development. Most of these relatively readily re-creatable habitats are typical of highly dynamic or relatively transient conditions: mudflats, saltmarshes, some fluvial communities, some saline lagoons and brackish conditions, and some freshwater wetlands. The results of a small number of inter-tidal habitat creation projects indicate that successful outcomes can be achieved, depending upon careful project design, consideration of morphological influences and sediment supply.

Even in apparently successful projects, a short-term inventory of species may not reflect the assemblage that can (or should) realistically be re-created. Plant and animal communities change over time, whether in response to the maturation of newly created habitats, or as a consequence of local or more widespread changes in land management or climate change.

A simple upward scaling of extent of habitat lost into extent of new habitat created cannot be seen as the answer in all cases to compensate for delays, lag-times and uncertainty over replicability, as used in port development cases. Where lost habitats are associated with highly mobile and dynamic environments such as estuaries or the open coast, or perhaps some fluvial systems, habitat re-creation at a scale greater than the area of prime habitat loss seems to be a realistic approach. Even so, there is sufficient variability in the confidence that can be placed on delivery of suitable compensatory measures that impacts must be addressed on a case-by-case basis rather than according to a simple formulaic approach. In the case of many terrestrial habitats there is greater uncertainty over the potential outcome of habitat creation and timescales needed for the habitat to mature are longer. Where differences between re-created and original habitat are particularly pronounced, much greater areas of new habitat will be needed to enhance the probability that the most vulnerable organisms have been accommodated. Even using such an approach, there is likely to be further deterioration of the overall stock of important biological interest, especially as there will almost certainly be lag times between the destruction of habitat and the creation of new habitat.

Even with relatively simple cases, data presented by Edgar et al. (2005) raise the important question as to the degree to which there can be confidence that mitigation or compensation measures will be delivered, monitored and reviewed. Where habitat creation is more difficult, the time-scales may not be practical in terms of legal reassurance that suitable replacement habitat has been secured. Furthermore, additional measures may be neces-

sary to respond to failure at some much later stage but they may be difficult to enforce.

Our review highlights many of the practical difficulties that have to be overcome if ancient semi-natural habitat is to be replaced by new habitat created as a compensatory measure. It is clear that the prospects for creating biologically comparable replacements that support an array of scarce and less mobile species are not encouraging, and any decision that involves such losses must therefore be the last resort. It therefore follows that in the majority of cases the concept of habitat banking is unlikely to confer particular advantages except perhaps within highly dynamic environments where recreation timescales are generally a matter of years rather than decades. We must therefore conclude that only where all alternatives have been explored and rejected for sound practical reasons should the creation of compensatory habitat be considered.

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