Rewilding - A New Paradigm for Nature Conservation in Scotland?

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Available online: 09 Mar 2012


To link to this article: http://dx.doi.org/10.1080/14702541.2012.666261

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Rewilding – A New Paradigm for Nature Conservation in Scotland?

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ABSTRACT Rewilding is a strategy for the conservation of complete, self-sustaining ecosystems, primarily involving the protection and, where necessary, reintroduction, of populations of keystone species in large, connected reserve networks. A potential method of preserving ecosystem functions and biodiversity, it is now receiving a great deal of practical and political attention, particularly in North America. In Scotland, where many native species have been extirpated in the relatively recent past, rewilding has clear relevance and may provide an overarching set of objectives for current programmes of native woodland restoration and species reintroductions. Nevertheless, rewilding is not widely used as a term or strategy in Scottish conservation. This review considers the development of the concept and its possible application in Scotland, and identifies substantial scope for rewilding, in terms of the restoration and protection of large areas of wild land, and of the reintroduction of native species which have been driven to extinction by human activity. As the environmental, social and economic benefits which are likely to result from a programme of rewilding in Scotland outweigh the potential drawbacks, the adoption of rewilding is recommended as one aim of environmental policy.

KEY WORDS: conservation policy, environmental restoration, keystone species, rewilding, Scottish environment, trophic cascades

Introduction

The howl of the wolf is perhaps the most potent and widely-recognised symbol of wilderness in the northern hemisphere. As a result of habitat loss and long-term persecution of wolves, it is rarely heard outside large areas which are relatively untouched by human activity. Along with other emblematic species such as the lynx and brown bear, the wolf has long been absent from even the most remote and wild corners of the Scottish Highlands. However, considerable potential exists here to restore habitats and extinct species through an approach termed ‘rewilding’. Despite having firm foundations in ecology, environmental philosophy and practical conservation, rewilding is yet to be widely adopted, as a term or concept, in...
Scotland. Nevertheless, its principles are highly applicable here and offer a valuable new perspective on Scottish environmental issues.

The formal theory of rewilding developed from the field of conservation biology in North America during the 1990s, and also has a longer and less distinct practical history in various countries including the UK. Its principal contention is that the conservation of biodiversity is best served through the protection of species at or near the top of the food chain in large, connected areas. This claim is justified through a number of established theories and, although controversial, has strong empirical support (e.g. Hobbs & Cramer 2008; Terborgh & Estes 2010).

Few investigations of the applicability of rewilding have been undertaken outside North America, especially where social or political conditions appear to preclude it. This is true in Scotland, where conservation is often allied instead to the concept of sustainable development, despite an independent practitioner-led rewilding movement influenced by evidence of extensive degradation of natural ecosystems and changes in the economic basis of current land uses. The focus of rewilding on the restoration of natural processes, along with its ultimate objective of establishing populations of large carnivores, may have a significant role to play in Scottish environmental management. In this article, we explore the philosophical and scientific foundations of rewilding and consider its relevance and meaning in Scotland.

**Historical Context**

Rewilding is often presented in the context of a long history of anthropogenic environmental disruption, the attempted reversal of which is one of its principal objectives. Undisturbed ecosystems are attributed inherent value, and human activity is regarded as inherently unnatural, to some extent at least (Soule´ & Noss 1998; Taylor 2005). Justifications for this rely on evidence of anthropogenic environmental damage throughout human history, beginning with the extinctions of numerous (mainly large animal) species from each landmass following human colonisation (Brook & Bowman 2002; Foreman 2004).

These extinctions have long puzzled biologists; to Darwin they represented a ‘most gratuitous mystery’ (1859, p. 341); to Alfred Russell Wallace, a ‘marvellous fact . . . that has hardly been sufficiently dwelt upon’ (1876, p. 150). Despite much investigation during the intervening years, their causes remain unclear. Human hunting has become the favoured explanation, but because colonisation is difficult to date confidently and was often, by its nature, coincident with climate change, it has proved difficult to link decisively to species extinctions in most cases (Hoffecker et al. 1993; Grayson & Meltzer 2003; Barnosky et al. 2004; Borrero 2008; Pushkina & Raia 2008).

Nevertheless, strong circumstantial evidence exists, particularly where colonisation was relatively recent. This is the case in the islands of Oceania, where up to 2,000 bird species, for example, became extinct within centuries of the arrival of humans (Steadman & Martin 2003; Sutton et al. 2008). Such findings have led to broad agreement that Pleistocene extinctions were most likely caused by some combination of human activity and climate change, perhaps exacerbated by other factors
With the development of arable agriculture c.10,000 BP, however, far more dramatic anthropogenic environmental change began to occur; its early effects can still be discerned along the waterways and coasts by which agricultural practices spread from the Middle East (Thapar 1978; Neumann 2003; Willcox 2004; Séféridès 2007; Shennan & Edinborough 2007; Atahan et al. 2008).

The development of the modern Scottish environment began at the end of the Pleistocene epoch, between 15,000 and 10,000 BP (Ballantyne et al. 1998; Currant & Jacobi 2001). In Scotland, the Holocene began with a predominant habitat of open tundra, which spread as the ice of the Devensian glaciation receded. This tundra quickly gave way to scrubland and woodland of pioneer species such as birch and, soon afterward, Scots pine (Froyd 2005). This flora gradually moved northwards along with mammals such as the Auroch, Beaver, Brown Bear, Lynx, Mountain Hare, Pika, Red and Arctic Fox, Red Deer, Reindeer, Saiga, Tarpan, Wolf, and Woolly Mammoth, although the extent to which some of these penetrated Scotland is unclear (Kitchener 1998; Yalden 1999).

Particular uncertainty surrounds human colonisation; the earliest evidence is found on the western coast and islands between 11,000 and 9,000 BP, contemporaneous with that from similar latitudes in Scandinavia (Edwards & Mithen 1995; Bang-Andersen 1996). It is thought likely that settlement occurred along the Atlantic seaboard between 10,000 and 8,000 BP, with the first definitive evidence coming from the island of Rum at 8,500 BP (Wickham-Jones & Pollock 1987; Wickham-Jones & Woodman 1998). Species present by the time of the formation of the English Channel c. 9,500–7,000 BP are now classed as native, and subsequent immigration was inevitably less intense. By the early Mesolithic, Britain contained an estimated 1.4 million Wild Boar, 100,000 Aurochs, 70,000 Elk, 35,000 Beavers, 20,000 Wolves, and 10,000 Lynx – some of the species that have since become extinct – alongside 2,000–3,000 humans (Yalden 1996, 1999).

Few environmental impacts are attributable to these people or their close descendants. A role in Machair formation, which began through natural processes in the Outer Hebrides 8,700 BP, has been theorised (Edwards et al. 2005a), and some intensive plant use bordering on agriculture may have taken place (Mithen et al. 2001). However, it was not until the Neolithic, beginning 6,500–5,000 BP, that significant anthropogenic change occurred. By this time, Scottish forests had reached their maximum extent, covering up to 80% of the country (Smout 2003). Their subsequent retreat under human pressure – a ‘vast tale of wanton destruction’ in the words of Frank Fraser Darling (Darling & Boyd 1964, p. 55) – provides the clearest evidence of changing land use. Once again the effects of simultaneous climate change, as Scotland became warmer and wetter, confuse the record and are thought by some to be solely responsible for widespread peat formation and the decline of forest cover (e.g. Fenton 2011). However, a clear pattern of woodland clearance emerged with the advent of agriculture, and continued for several millennia (Smout 1997, 2003).

Evidence of moderately intense agricultural activity in the Bronze Age (c. 4,500–2,500 BP) is found throughout Scotland, but especially in the Western and Northern Isles (Edwards et al. 2005b; Tipping et al. 2008). Woodland area declined steadily as a result, with arable and pastoral land being created through burning and clearing.
(e.g. Wilkins 1984). The Iron Age (c. 2,500–1,500 BP) saw an escalation of these practices, especially in the south (Davies 2007; Dumayne-Peaty 1998, 1999).

This large-scale forest clearance contributed to the first Scottish faunal extinctions that are clearly attributable to man – those of the Elk and the Lynx (Table 1). The Elk survived in Scotland until at least 4,000 BP, and the Lynx until at least 1,800 BP (and even more recently in England). It is likely that both species persisted for some time after these dates, but more recent remains have not been found. In any case, the climate was by then settled and similar to that of the present time, and species were primarily threatened by human hunting, habitat destruction and persecution, particularly when they posed a threat to domesticated livestock, as in the case of Lynx (Yalden 1999; Hetherington et al. 2005).

The Auroch (which became globally extinct in 1627) probably succumbed to habitat loss and hunting during the Neolithic, but may have been present throughout the Iron Age and perhaps into Roman times. The latest certain British specimen was found in England and dates to 3,200 BP (Kitchener 1998; Yalden 1999). The brown bear is thought to have survived until the 10th century (1,000 BP), although the most recent confirmed remains date to 2,700 BP (ibid.). Prior to its extinction, it was hunted (perhaps as a game animal) and used in Roman gladiatorial contests (Harting 1880; Yalden 1999).

Medieval Scotland was characterised by social, rather than environmental, change. Interactions between Scandinavian, Gael, Briton, Anglo-Saxon and Norman populations intensified, with dramatic effects (McDonald 1995). Coastal regions, particularly the west and the islands, became increasingly important as trading hubs (Barrett et al. 1999). Inland, diffuse settlement and stock grazing spread rapidly (Dodgshon 1993; Holl & Smith 2007). Conflict was the chief driver of environmental

**Table 1. Scottish faunal extinctions of the Holocene directly attributable to man**

<table>
<thead>
<tr>
<th>Species</th>
<th>Date of extinction</th>
<th>Probable causes of extinction*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Elk</td>
<td>&lt;4,000 BP</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Auroch</td>
<td>~2,000 BP (?)</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Lynx</td>
<td>&lt;1,800 BP</td>
<td>Hunting, persecution, habitat loss</td>
</tr>
<tr>
<td>Brown Bear</td>
<td>10th century (?)</td>
<td>Hunting, persecution, habitat loss</td>
</tr>
<tr>
<td>Crane</td>
<td>&lt;15th century (?)</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>White Stork</td>
<td>15th century</td>
<td>Habitat loss</td>
</tr>
<tr>
<td>Beaver</td>
<td>16th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Great Bustard</td>
<td>16th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Wild Boar</td>
<td>17th century (?)</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Wolf</td>
<td>17th century</td>
<td>Hunting, persecution, habitat loss</td>
</tr>
<tr>
<td>Capercaillie</td>
<td>18th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Great Auk</td>
<td>19th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Bittern</td>
<td>19th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Red Squirrel</td>
<td>19th century (?)</td>
<td>Habitat loss</td>
</tr>
<tr>
<td>Great Spotted Woodpecker</td>
<td>19th century</td>
<td>Hunting, habitat loss</td>
</tr>
<tr>
<td>Red Kite</td>
<td>20th century</td>
<td>Persecution</td>
</tr>
<tr>
<td>Goshawk</td>
<td>20th century</td>
<td>Persecution</td>
</tr>
<tr>
<td>Spotted Crake</td>
<td>20th century</td>
<td>Habitat loss</td>
</tr>
<tr>
<td>Polecat</td>
<td>20th century</td>
<td>Persecution</td>
</tr>
<tr>
<td>Osprey</td>
<td>20th century</td>
<td>Persecution</td>
</tr>
<tr>
<td>Sea Eagle</td>
<td>20th century</td>
<td>Persecution</td>
</tr>
</tbody>
</table>

*Principal suspected causes of extinction are given (Kitchener 1998; Yalden 1999). ‘?’ denotes dates over which there is particular uncertainty.
change, however, with extensive burning of woodland in the north and west (especially by the Vikings) and from the south-east (by various warring factions) (Darling & Boyd 1964).

The resulting loss of habitat, together with over-hunting, led to the extinction of the Beaver during the 16th Century (Kitchener & Conroy 1996). The White Stork and, possibly, the Crane had already succumbed to habitat loss; the Great Bustard followed (Kitchener 1998). The Wild Boar is thought to have died out in the early 17th Century for similar reasons, although the species persisted as domesticated stock and was repeatedly reintroduced to English game forests (Ritchie 1920; Yalden 1999). Destruction of woodland was also detrimental to Wolf populations and, after centuries of persecution had ‘failed to compass their entire destruction’, they were ‘effectively extirpated’ by ‘cutting down or burning whole tracts of the forests which harboured them’ (Harting 1880, p. 116). This almost certainly occurred during the 17th Century, despite the legendary killing of Scotland’s last wolf in 1743 (Darling & Boyd 1964; Yalden 1999).

By the mid-18th century, only around 4% of Scotland was wooded (Stewart, in Smout 2003). Human populations, grazing (which intensified dramatically with the advent of large-scale sheep farming) and industrial activity all increased during the next century, while the failed Jacobite Rebellion of 1745–1746 led to yet more woodland burning and dramatic social change. The clearance of tenant crofters and farmers from Scottish estates remains an infamous episode from this period, and marked a shift away from the diverse if intensive environmental management practiced by these people, towards extensive grazing regimes (Smout & Watson, in Smout 1997; Davidson 2004).

The Red Squirrel was almost or entirely lost from Scotland during this time (but was quickly reintroduced) (Yalden 1999). Many bird species also fell victim to habitat loss and hunting – the Capercaillie in the late 18th Century; Bittern, Great Auk, and Great Spotted Woodpecker in the 19th; and the Spotted Crake in the early 20th (Bourne 1972; Kitchener 1998).

Sporting estates proliferated during the 19th century and used reintroductions to bolster the populations of species which had acquired new value as game, perhaps even saving some from extinction. These included the Red Fox (even as attempts were made to extirpate it elsewhere), Otter, Red and Roe Deer (Yalden 1999; Fairnell & Barrett 2007). Many other species were less fortunate, however - particularly those classified and intensively persecuted as vermin. Species such as the Pine Martin and Wild Cat suffered enormous declines, almost following the Polecat to extinction (Hubbard et al. 1992). The Osprey, White-Tailed Sea Eagle, Red Kite and Goshawk were also lost (Kitchener 1998).

Philosophical and Scientific Basis for Rewilding

By this point, the development of the modern conservation movement in the United States of America was well underway. Its origins are often linked to the émigré Scotsman John Muir, who would become one of the foremost campaigners for the protection of North America’s apparently pristine environments. Although environmental restoration was a long-established practice in western Europe, it often focused on preserving the anthropogenic, mainly agricultural (often referred to
as ‘cultural’) landscapes that had developed over millennia of relatively consistent land use (Hall 2005). The long-term moulding of nature to suit the purposes of arable and pastoral agriculture had left relatively few areas of true wilderness, and the protection or enhancement of these was a small, if important, facet of European conservation. The ‘New World’, in contrast, contained environments that appeared untouched to European colonists, suggesting a new paradigm for environmentalism in which the effects of human activity were to be actively reversed (ibid.; Abensperg-Traun et al. 2004).

Muir was a leading proponent of the preservationist view, which achieved prominence with the designation of Yosemite National Park in 1872 and emphasised the aesthetic and spiritual (and hence largely anthropocentric) value of spectacular natural areas (Runte 1987; Muir 1996). At around the same time, the Progressive Movement helped to popularise a concept of the environment as a common resource requiring protection for the benefit of all (Scott 1959; McDonagh 1992) – later labelled as ‘limited socialism in the public interest’ (Bates 1957, p. 30). Those most associated with this concept include Republican President Theodore Roosevelt and the first Chief of the US Forest Service, Gifford Pinchot (Meyer 1997; Miller 2004), who argued that ‘the first principle of conservation is development, the use of the natural resources . . . for the benefit of the people’ (Pinchot 1910, p. 43).

The preservation of natural resources for their ongoing exploitation was already an established practice both within North America and beyond. This was particularly true of forestry, wherein ‘sustained yield’ approaches provided the basis for management on both sides of the Atlantic from the early 19th century (Farrell et al. 2000; Papaik et al. 2008). In Scotland, sporting estates also traditionally practiced resource management of woodlands and, increasingly, species classed as game. Beyond such public and private approaches to resource management, the concept of common ownership of, and rights to, environmental resources has been a fundamental aspect of many indigenous cultures around the world (Berkes 1999; Colding & Folke 2001; Campbell & Butler 2010). This has often resulted in deliberate attempts to achieve sustainable relationships with the environment, as apparent in traditional patterns of resource use and recognised in conceptions of Traditional Ecological Knowledge (Pierotti & Wildcat 2000, Turner et al. 2000; Davis & Ruddle 2010). In late 19th century North America, however, the prominence afforded to the idea of common ownership of environmental resources by Pinchot was far more unusual.

Equally influential was the emerging field of ecology, famously given a moral element in Aldo Leopold’s ‘land ethic’ (Leopold 1966). This championed the inherent value of ecological systems, and contributed to the establishment of ecological restoration as a practical process (Hall 2005). Notably, it also stated that humans are integral members of the ecological community (Skolimowski 1984; Taylor 1991). Once again, this was extensively presaged by belief systems around the world which represented humankind as one member of an inter-reliant ecological system (Egerton 2001; Swearer 2001; Watson et al. 2003; Martin et al. 2010). These systems were often geographically delimited, for example corresponding to specific watersheds (Berkes et al. 1998). Leopold’s philosophy had more immediate roots in the burgeoning environmental and Deep Ecology movements, typified by the writings of Arne Naess, Rachel Carson and Paul Ehrlich (e.g. Carson 1962;
Ehrlich 1968; Naess 1973). In bridging the gap between scientific and ethical fields of ecology, however, it proved particularly influential.

Scientifically, ecology led to a repudiation of protection on purely aesthetic grounds, promoting biological significance as an alternative focus for conservation effort (Soule 1985; Kuzmiak 1991). Biological conservation subsequently developed as an attempt to maintain diversity, initially stressing the accepted importance of wilderness areas and the presence of large carnivores within them (e.g. Shelford 1933). Practical efforts became increasingly targeted at specific species, habitats or areas, however – exemplified in the controversial identification and protection of global biodiversity ‘hotspots’ (Olson et al. 2001; Bartolino et al. 2011; Thomassen et al. 2011). The consequent risk of habitat fragmentation, genetic impoverishment and species loss was a motivating factor in the development of rewilding (Macarthur & Wilson 1967; Hanski & Gilpin 1991).

Keystone Species

Of equal importance to rewilding was the development of the theory of keystone species. These are defined as species which have a structural role in their community disproportionate to their abundance, and are often large, wide-ranging predators (e.g. Power et al. 1996). The putative ‘regulatory role of large predators’, based on this theory, is central to the concept of rewilding. Here it is presented in a tripartite form in which such species suppress herbivores through predation, suppress smaller mesopredators through competition, and consequently limit competition between prey species (Soule & Noss 1998).

While proponents of rewilding regard these effects as proven (Soule & Noss 1998) debates about their nature and extent continue (Paine 1995; Isasi-Catala 2011). Nevertheless, strong empirical support for the existence of keystone species does exist. The maintenance of diversity through predation is implied by Gause’s (1934) competitive exclusion principle, which follows Darwin (1859) and Grinnell (1917, 1924) in suggesting that individuals best fitted to their biotic and abiotic environment are able to drive competitors to extinction in the absence of some mediating process. Predation upon these superior competitors may fulfil this role, so enabling a greater number of species to persist (Slobodkin 1961, 1964). This was confirmed by Paine (1966), who found that ‘local species diversity is directly related to the efficiency with which predators prevent the monopolization of the major environmental requisites by one species’ (p. 65), and subsequently labelled such predators ‘keystone’ in respect of their stabilising influence on community function (Paine 1969).

In the intervening years, similar effects have been observed in a wide range of ecological communities. In tropical rainforests – some of the most diverse ecosystems on Earth – plant resources which sustain animal populations may be regarded as keystone (Terborgh 1986), as may animal or plant species upon which others depend (Hubbell & Foster 1986). The search for keystone species in these and other environments has been intensive (e.g. Wright et al. 1994; Peres 2000; del Castillo et al. 2009, Jordán 2009).

An important implication of the theory of keystone species is that species removal may result in a ‘trophic cascade’ of further species loss (Terborgh & Estes 2010). This too has been observed on numerous occasions (ibid., Crooks & Soule 1999;
Schmitz et al. 2000; Jackson et al. 2001; Shurin et al. 2002; Ripple & Beschta 2008), particularly where forest fragments or islands have been left devoid of predators (Terborgh et al. 2001). Conversely, ecosystem function can recover following the return of keystone species. After wolves recolonised Banff National Park in Canada in 1986, cascading effects were identified on elk survival rates and population density, aspen and willow recruitment, beaver density, and riparian songbird diversity and abundance (Jones 2002; Hebblewhite et al. 2005). Similarly, the programme of wolf reintroduction to Yellowstone National Park had dramatic effects on the population and movement of elk, leading to the recovery of aspen, beaver and riparian habitats (Ripple & Beschta 2003, 2006, 2007; Beschta & Ripple 2010). Conditions following these changes appear to approximate conditions prior to wolf extirpation (Halofsky & Ripple 2008). Other ecosystems have shown a similar recovery of prior structure (Mittelbach et al. 1995; Hobbs & Cramer 2008), and smaller-scale equivalents are found with variation in predator density (e.g. McLaren & Peterson 1994). It has also been argued that biotic resources and processes which benefit humans (known as ecosystem services) are enhanced by the presence of keystone species; these may include the provision of clean air and water, pollination of agricultural crops, nutrient cycling and storage, and the mitigation of the ecological effects of climate change (Hooper et al. 2005; Fischer et al. 2006; Larson & Paine 2007).

The protection of keystone species may have additional indirect effects. Many such species are wide-ranging and sensitive to disturbance, requiring large, relatively pristine habitats which also support many other species. The conservation of such species can therefore have an ‘umbrella’ effect of conferring wider indirect protection. The efficacy of this approach does remain under debate, however (Lambeck 1997; Roberge & Angelstam 2004; Isasi-Catala 2011).

Single species may also be used to justify the connection of protected areas in order to provide space for viable populations (Hilty et al. 2006). Other potential benefits of this include increases in diversity and new opportunities for economically productive land uses (Kerley et al. 2003; Chave & Norden 2007). While debate continues over the design and efficacy of corridors for particular species (Haddad et al. 2003; Van Dyck & Baguette 2005), these general effects are increasingly well-documented (Gilbert-Norton et al. 2010). The establishment of connections between protected areas has consequently been identified as a global priority for conservation (Crooks & Sanjayan 2006; Worboys et al. 2010).

Theory of Rewilding

The first detailed definition of rewilding traced its development through Muir’s monumentalism and the associated ‘wilderness movement’, the science of biological conservation and targeted species protection, and the discoveries of island biogeography (Soule` & Noss 1998). Foreman (2004) highlights the precedent of ecological restoration in identifying the need – if not the means – for restoring ecosystem health. Soule` and Terborgh (1999) argue that ecological restoration is more concerned with small-scale recovery of ecological process, while Simberloff et al. (in Soule` and Terborgh 1999) further contend that it requires, by definition, an implicit acceptance of a prior state to which restoration is directed. The selection of
this prior state, they argue, is ‘fundamentally normative rather than scientific’ (p. 65). Rewilding differs in that its target – the presence of keystone species – is ‘both an end (because of our duty to repair past mistakes in management) and a means by which the viability of conservation units is achieved’ (Soule and Terborgh 1999, p. 11).

Soule and Noss (1998) also suggest that rewilding has a strong moral justification linked to Aldo Leopold’s land ethic, under which the deliberate and often officially sanctioned extirpation of species must be reversed. This commitment is extended by others to include species unintentionally eliminated by human action and other anthropogenic environmental damage, albeit in relation to the conceptual focus on keystone predators (e.g. Foreman 2004; Taylor 2005). Aesthetically, wilderness is linked with the presence of large predators, echoing Muir’s assertion that ‘none of Nature’s landscapes are ugly so long as they are wild’ (Muir 1901, p. 4). Many treatments of rewilding give this as its principal extra-scientific rationale despite its subjective nature (e.g. Meyer 1997; Noss et al., in Soule´ & Terborgh 1999; Trombulak & Royar, in Klyza 2001; Foreman 2004; Taylor 2005).

In Britain, a less conspicuous but largely independent rewilding movement developed through practical conservation and restoration projects (Featherstone 1997; Ethos 2008; Taylor 2011). Partly as a result, it is avowedly more loosely defined and less clearly differentiated from other restorative strategies; something that is itself regarded as a defining characteristic (Taylor 2011). Nevertheless, it shares the overarching aim of protected and connected ‘core’ areas of wild land – inspired here by the practicalities of conservation in a densely populated island nation – with the North American concept, differing most clearly in the initial steps required for the reintroduction of species (Featherstone 2004; Ethos 2008). A spiritual justification is more often employed by British than North American commentators – ironically based to some extent on Native American environmental philosophy (e.g. Taylor 1996, 2005; Featherstone 1997).

Pinchot’s philosophy of resource conservation is less often employed, and direct socio-economic returns of rewilding generally remain unspecified. This contrasts with the extensive literature relating to the effect on ecosystem services of other conservation strategies (e.g. Baron et al. 2002; Lomas et al. 2008; Norgaard & Jin 2008). Whether this stems from an ethical unwillingness to entertain an anthropocentric case for rewilding or from uncertainty that the case could be made is unclear, and may vary between commentators. Nevertheless, greater consideration does appear to have been given to the economic consequences of rewilding in the British movement than the North American (e.g. Taylor 2005, 2007).

In North America, according to Soulé and Noss, ‘the greatest impediment to rewilding is an unwillingness to imagine it’ (1998, p. 26); in Britain, Taylor similarly argues that ‘as with all paradigm shifts . . . the most resistant force . . . is bureaucratic’ (2009, p. 1). However, these authors also acknowledge practical problems in establishing wilderness areas, particularly in populated or highly degraded regions. Soulé and Noss suggest that socioeconomic objectives cannot be incorporated in a realistic conservation plan; a potentially problematical contradiction of sustainable development theory and of suggestions that ecological restoration should incorporate social and cultural interests (e.g. Robinson 2011). Taylor, in contrast, identifies the lack of natural vegetation and grazing regimes in Britain as a greater difficulty, requiring remedial steps to be taken in preparation for full rewilding.
Further difficulties relate to reintroductions. The opinions of both the public and landowner have been found to have a decisive effect (perhaps equal to biological factors) on the feasibility of reintroduction programmes, to which they are often opposed (Clark et al. 2002; Ostergren et al. 2008; Worthington et al. 2010). Keystone species are often viewed especially negatively and persecuted as ‘pests’ or ‘vermin’, despite their roles in sustaining ecosystem functions from which people benefit both directly and indirectly (Delibes-Mateos et al. 2011). This has been successfully addressed in some countries, however, through informed discussion and a legal framework to provide justification and compensation for losses sustained (Dyar & Wagner 2003; Meadow et al. 2005).

Nevertheless, more fundamental objections to the theory can be made. The ethical case for rewilding may be viewed as subjective in its emphasis on moral obligations to repair anthropogenic environmental damage over moral obligations to provide for expanding or impoverished human populations (West & Brechin 1991; Fabricius et al. 2004), although proponents argue, as above, that it is not its function to stress the latter. Attempts to balance the two have been made in a Scottish context and inform the Scottish Government’s attempted amalgamation of environmental and socio-economic objectives in a programme of sustainable development (Scottish Government 2011). The claim that rewilding can achieve both by producing healthier and more robust ecosystems (e.g. Daily 1997) relies on empirical evidence that, while strong, is not conclusive. In particular, reintroductions of species identified as keystones have occurred so rarely that their detailed impacts on environmental and socio-economic conditions remain hard to predict; a significant difficulty for a theory that places so much emphasis on them.

Conservation Philosophies and Environmental Realities in Scotland

Whilst the foundations of rewilding were being laid in North America, the Scottish environment was subject to continuing changes in human land use. Much of the 20th century was characterised by exploitative practices and attitudes. These were perhaps most evident in the management of Scottish woodlands after the First World War, when demand for timber led to widespread felling and to the establishment of the Forestry Commission in 1919 (Dunlop, in Smout 1997). Within 20 years, almost 200,000 ha of commercial forestry – mainly comprising exotic species – had been planted in Scotland (Holmes 1975). Little of this ‘strategic reserve’ had reached maturity by the time of the Second World War, and so the bulk of the 92,000 ha felled between 1939 and 1945 comprised pre-existing forest (Foot, in Smout 2003).

At the same time, conservation movements were developing throughout the UK. The ‘environmental revolution’ of the 1960s affected attitudes in Britain, but the increasing appeal of conservation here may be more accurately viewed as part of the legacy of the Romantic reaction to industrialisation (Kuzmiak 1991; Veldman 1994). The advent of widespread car ownership and air travel, which greatly increased access to remote and natural areas, may also have stimulated public awareness of environmental issues (Bloom 1995; Turner 2002). Membership of environmental groups grew significantly and a range of statutory and non-governmental advisory and advocacy groups were established (Adams 1997).
Another important current in British environmentalism was—and remains—respect for agriculture. As elsewhere in Europe, a long history of agricultural land use has generated widespread association of agricultural objectives with social, and even environmental, benefit (Bills & Gross 2005; Soliva et al. 2008). In Highland Scotland, the legacy of the ‘Clearances’ has bolstered the perceived cultural significance of the crofting system, and prompted its statutory protection (Moisley 1962). Together with domestic and European monetary support such as that generated by the Common Agricultural Policy (CAP), and a prevailing anti-regulatory political climate, these factors have ensured that agriculture has become increasingly associated with—yet often detrimental to—the British natural environment (Donald et al. 2001). This has motivated a number of policy reforms, including changes to the CAP designed to favour less intensive and more diverse agricultural practices, exemplified by the introduction of the Single Farm Payment scheme in 2005 (Europa 2009; Scottish Government 2010b).

In upland Scotland, the principal form of agriculture since the 18th century has been sheep farming (Dodgshon & Olsson 2006). In recent years, however, subsidies to the industry have declined, and the area under sheep grazing fell by over 1 million ha between 2004 and 2009, to just under 7 million ha excluding common grazing. The average net farm income reached a low in 2006/07 of just £1,932, while average subsidies of £23,702 were maintained by the classification of 84% of Scotland—and over 90% of grazing land—as agricultural ‘Less Favoured Areas’ (Scottish Executive 2006, 2007a,b; Scottish Government 2008, 2010a). Scotland does have populations of native herbivores, and low- to moderate-intensity grazing can have demonstrable environmental benefits, particularly in terms of floral diversity (Humphrey & Patterson 2000; Marriott et al. 2009; Whyte 2010). This is especially true of large native herbivores such as wild cattle and wild boar (Armstrong & Bullock 2004; Hancock et al. 2010). However, no grants exist to support the establishment of natural grazing regimes (Taylor 2009), while intensive grazing of the kind often produced by modern farming practices is highly detrimental to diversity and ecological function (Dennis et al. 2008; Newton et al. 2009).

The majority of Scottish estates are managed for sport hunting, particularly of deer and grouse. The economic basis of these industries is also precarious, and estates generally require financial support by their owners—608 of whom possess half of Scotland’s land in estates typically of 5,000–8,000 ha, representing the highest concentration of large-scale private land ownership in the world (Wightman 1999; Price et al. 2002; MacMillan et al. 2010). A major consequence of the estate system is high densities of deer: over the past 30 years, it is estimated that deer numbers in Scotland have doubled, to well over 300,000 (Clutton-Brock et al. 2004; Nilsen et al. 2007). Increased levels of culling in recent years have resulted in red deer populations stabilising or declining in certain areas, but their grazing and browsing (particularly when combined with that of sheep) continue to have detrimental localised impacts on upland ecosystems, particularly through the inhibition of semi-natural native woodland regeneration (Côte et al. 2004; Albon et al. 2007). Management for grouse shooting appears to have mixed effects depending upon its nature, maintaining large areas of heather moorland (some of which represent internationally significant habitats) and benefitting certain species as a result, but also preventing natural reforestation and proving detrimental to other species, particularly those which
predate on grouse and are therefore often persecuted (Thompson et al. 1995; Thirgood et al. 2000; Tharme et al. 2001) However, economic pressures and changes in estate-owner motivations are leading to diversification in land use, with several estates conducting conservation or restoration projects (e.g. The Corrour Trust 2010; Alladale Wilderness Reserve 2011).

The unique history and structure of land use in Scotland means that conservation must take place alongside other forms of active management. The philosophical basis of conservation is consequently rather different to that in North America, as reflected in the emphasis of conservation designations. The UK’s basic form of statutory protection, the Site of Special Scientific Interest (SSSI), is intended to ‘conserve the total national . . . range of variation in habitats, with their associated flora and fauna’ (JNCC 2011b, p. B-1.2), and is applied to almost 13% of Scotland’s land area in 1,456 different sites (Joint Nature Conservation Committee (JNCC) 2011a; Scottish Natural Heritage (SNH) 2011b). However, where potentially damaging activities are proposed within sites, Scottish Natural Heritage must balance environmental considerations against ‘the needs of agriculture, fisheries and forestry; social and economic development; . . . and the specific interests of owners and occupiers and local communities’ before giving or withholding consent (SNH 2004). National Parks, meanwhile - the first of which was designated in 2002 – aim, in part, to ‘promote the sustainable use of the natural resources of the area . . . [and] sustainable social and economic development of the communities of the area’ (SNH 2011c). This differs notably from US National Park objectives to ‘preserve unimpaired . . . natural and cultural resources . . . for the enjoyment, education, and inspiration of this and future generations’ (National Park Service, NPS 2011).

In these respects, something like Pinchot’s natural resource ethic is recognisable in Scottish environmental thinking, with protection generally provided on the basis of ongoing resource use, albeit ‘sustainable’. What appears to be lacking is an underlying principle of equitable distribution. There is, however, an independent Scottish tradition of universal access to environmental benefits, apparent in the convention of ‘open access’ to the countryside which became a legal right under the Land Reform (Scotland) Act 2003. In fact, practical conservation projects in Scotland often have more in common with rewilding theory than official management policies suggest, even avowedly eschewing anthropocentric justifications in favour of those that assign inherent value to wild land (e.g. National Trust for Scotland 2002; Carrifran Wildwood 2011; Trees for Life 2011).

A strong aesthetic tradition also informs conservation in Scotland. While agricultural landscapes are valued, those perceived as ‘wild’ are attributed even greater significance, with the great majority of the Scottish public regarding their presence and protection as very important (McMorran et al. 2006; SNH 2008). They are consequently recognised in Scottish Government policy (SNH 2002; Scottish Government 2009) and in the objectives of several conservation charities concerned with land management (e.g. Carrifran Wildwood 2011; John Muir Trust 2011; Trees for Life 2011). General agreement also exists that, although subjective, the phrase ‘wild land’ implies a lack of human artefacts (McMorran et al. 2008). Furthermore, the mapping of wildness as a landscape characteristic, using GIS techniques, has taken place in both of Scotland’s national parks and is
underway at the national level by Scottish Natural Heritage (Carver et al. 2008, 2012).

Despite this recognition of the importance of wild land in Scotland and a growing emphasis on species reintroductions, neither Scottish Government nor Scottish Natural Heritage policy statements refer to the concept of rewilding. In addition, the landscape-based Scottish concept of ‘wild land’ has yet to be meaningfully linked with large-scale approaches to ecological restoration or rewilding, although recent reports to the Scottish and UK governments recommend policies directed to this end (Fisher et al. 2010; Lawton et al. 2010). This is likely a consequence of the management-oriented approach to conservation in Scotland, and a general perception that rewilding is inapplicable here due to environmental and political conditions. Perhaps for similar reasons, an intrinsic land ethic is also apparently lacking in Scottish conservation theory, despite being afforded great significance by some practitioners and commentators (Featherstone 1997; Taylor 2005). It is perhaps most discernible in scientific arguments for biodiversity conservation, but normally remains implicit and so, at best, indefinitely and subjectively understood (Fischer & Young 2007).

The Potential and Justification for Rewilding in Scotland

The scientific justification for rewilding may be expected to have universal applicability. The hypothesised regulatory role of large predators and predicted and observed trophic cascades are not site- or country-specific and may be viewed as naturally-occurring processes in Scotland. Moreover, herbivores in Scotland, free of all large natural predators, are present in densities more than capable of supporting viable populations of all extirpated native carnivores (Wilson 2004; Hetherington & Gorman 2007). Scotland does differ from North America in the extent of remaining habitat available to such species. Nevertheless, studies have found ample scope for many missing species.

Beaver reintroduction is seen as feasible, and is the subject of an ongoing trial (South et al. 2000; Vines 2007; Scottish Wildlife Trust 2010). It can also be expected to generate considerable economic and environmental benefits, particularly in terms of diversity in riparian habitats (Campbell et al. 2007; Rosell et al. 2005). A viable population of Wild Boar could survive in Scottish woodland – as one does in southern England – and this species is also the subject of a trial reintroduction (Leaper et al. 1999; Trees for Life 2009). Scotland could also support a successful and relatively large population of Lynx (Hetherington & Gorman 2007; Hetherington et al. 2008), and a smaller but viable number of wolves (Yalden 1986; Panaman 2002; Nilsen et al. 2007). Both species would have significant effects on deer populations and movement, likely generating beneficial trophic cascades similar to those observed in Yellowstone and elsewhere (Johnson 2010; Melis et al. 2010).

Furthermore, Yalden (1986, 1999) has proposed that Aurochs and Tarpan could be ‘reconstituted’ from existing wild and semi-wild horse and cattle populations. Their reintroduction would be relatively simple, and beneficial to natural woodland habitats (Kirby 2004). Earlier successful reintroductions of Capercaillie and Sea Eagles could be repeated if necessary (as they might be for Capercaillie given high
numbers of fatalities in deer fences) (Love & Ball 1979; Moss 2001; Marshall & Edwards-Jones 2004). One exception is the Brown Bear, for which insufficient habitat is probably available at the present time, and which is not thought to play a keystone role (Taylor 2005).

Another consideration in Scotland is that many relevant species were eradicated several hundred years ago, so that ecological relationships between surviving species may have become altered in their absence. Such timescales may allow for some ecological and evolutionary change, but are unlikely to have resulted in transformed ecological relationships (Callicot 2002; Carroll et al. 2007). In addition, the well-documented recovery of varied environments following decreases in grazing intensity in Britain and elsewhere suggests that prior ecological structure may be spontaneously re-established (e.g. Welch & Rawes 1964; Virtanen et al. 2002; Bowen et al. 2007).

While interventionist management techniques would be necessary to implement rewilding, these do not necessarily differ from those already in use for species reintroductions and habitat restoration (e.g. Trees for Life 2009; Scottish Wildlife Trust 2010), or designated area management (SNH 2011b,c). The principal Scottish upland land uses which are not connected to conservation also require extensive management, be it for grouse shooting, forestry and woodland management, deer stalking or sheep farming (Thirgood et al. 2000; Price et al. 2002). As such, the great majority of uplands in Scotland are under ongoing human management at a level which rewilding may initially match, but would only decrease in the long term. The implied preclusion of other conservation projects in core areas may be viewed as a disadvantage, but this stems partly from their expected redundancy in healthy ecosystems and partly from scepticism about the efficacy of species-specific approaches to conservation (Soule et al. 2005; Early & Thomas 2007).

Not only is rewilding possible in Scotland, but it may be expected to provide numerous ancillary benefits. Perhaps the most obvious of these is the further development of nature-based tourism, which is currently worth at least £1.4 billion annually to the Scottish economy, and supports 39,000 full-time jobs (Bryden et al. 2010). Tourism associated with Sea Eagles on the island of Mull generates an estimated £2 million annually; dolphin-watching off the east coast another £4 million (ibid.). Reintroductions could be similarly profitable: it has been estimated, for example, that a beaver release site could be worth £2 million to the local economy (Campbell et al. 2007). The identification of scenery as the principal attraction for tourists who visit Scotland (VisitScotland 2008) further suggests that the diverse and natural landscapes produced by rewilding could be an important economic resource. Research in Scotland also suggests that local economic benefits of rewilding projects, including the number of full-time jobs sustained, are up to 5 times greater than those of traditional land uses (Taylor 2007).

Precedents exist in other countries for the economic benefits of rewilding, where restored habitats and wild populations of species such as beaver, lynx and wolf add dramatically to tourism revenue. The reintroduction of wolves to Yellowstone National Park, for example, resulted in a 25% increase in visitor numbers within three years, bringing estimated extra revenue to the local economy of $35.5 million annually – offset only by depredation losses of between $11,000 and $64,000 (Switalski et al. 2002; Duffield et al. 2006). The very act of protecting lynx habitat in
Canada has also been found to have a net positive economic effect (Kroeger & Casey 2006). Scotland’s hunting industry, itself worth an estimated annual £136 million (Bryden et al. 2010), also stands to gain. The diversification of game hunting to species such as elk, lynx and boar is already occurring profitably in other European countries, and improved habitats would allow current game species such as deer to command higher prices (Matilainen & Keskinarkaus 2010).

General environmental benefits would also result from rewilding. Considerable evidence suggests that protection of large areas – especially at the scale of entire ecosystems – is the most effective method of preserving biological diversity, especially under climate change (Hilty et al. 2006; Cantú-Salazar & Gaston 2010; Rands et al. 2010). Not only is this a global priority (United Nations 2011), but the Scottish Biodiversity Strategy explicitly advocates linked and large-scale habitat conservation to this end (Scottish Executive 2004). In England, the protection and connection of fragmented environments has been identified as the best method of ensuring ecosystem resilience and stemming declines in biodiversity (Lawton et al. 2010). This is reflected in recent UK Government policy which aims to expand and link protected areas in order to mitigate the effects of climate change on native species (HM Government 2011).

Efforts to create links between areas of wild land in Scotland may require greater recognition of core wild areas and of existing or potential wild corridors between them. A recently published report commissioned by the Scottish Government seeks to address these issues by considering policy responses across Europe, and suggests that clearly defined, zoned and linked protected areas of the kind required under rewilding are indeed necessary for long-term conservation (Fisher et al. 2010). Some designations, such as those of the International Union for Conservation of Nature (IUCN), already capture much of Europe’s wildest land and are highly compatible with the diverse environmental and socio-economic consequences of rewilding (ibid.). Specifically, categories Ia, Ib would be suitable for large ‘core’ areas, category IV for areas under restoration, and categories II, V and VI for buffer and surrounding regions where increasing levels of human activity occur. The IUCN has already initiated an expansion of the coverage of its protected-area management categories across the UK (International Union for Conservation of nature (IUCN) 2011). Emergent national maps of wildness in Scotland developed by SNH represent a potential starting point for the strategic recognition and protection of core wild areas and corridor zones (see Carver et al. 2008, 2012).

The potential of rewilding to increase the resilience of Scottish species and ecosystems to climate change would also have important consequences for the provision of ecosystem services (e.g. Lawton et al. 2010). A recent assessment of these underlines their importance, identifying them as including natural resources such as grass, timber, fruit, meat and water; processes such as carbon storage, protection from erosion and floods and the provision of clean air; and recreational opportunities such as wildlife watching, hunting, fishing and walking (SNH 2011a). Finally, rewilding satisfies the frequently-identified moral obligation to reverse anthropogenic environmental damage – particularly that which has been caused deliberately, as in the case of targeted species extirpations (Rolston 1985; Foreman 2004). This is recognised in British and European law, under which the Scottish
Government is committed to considering the reintroduction of extinct native species (Council of Europe 1979; Rees 2001).

Conclusion

In North America, the concept of rewilding was born of the precipitate environmental change generated by European colonisation of the continent and the strongly preservationist conservation movements that resulted. In the UK, a convergent yet largely independent movement grew from practical restoration projects which highlighted the far more gradual, though fundamental, ecological effects of human activity. Despite their different backgrounds, both formulations agree on the necessity of protecting large, connected areas within which self-sustaining populations of keystone species may survive.

In practice, rewilding must be a longer-term and initially less ambitious strategy in countries which lack a large proportion of their native large mammal species and natural habitats. Neither the British nor North American concepts, however, limit the methods by which their objectives may be achieved. Indeed, the principal virtue of rewilding may be its identification of a clear philosophical and scientific objective, often only implicit in alternative strategies, which does not require didactic and site-specific definition. As such, it represents a logical extension of the principles of nature conservation and an important regulative ideal.

Rewilding could therefore make an invaluable contribution to conservation philosophy and practice in Scotland. Its implementation here would be based on a coherent and scientifically defensible series of steps that would quickly provide wider benefits. In order to initiate this, the following key overarching aims are proposed for the adoption of rewilding as a conservation strategy in Scotland:

- The inclusion of rewilding as a term and objective in government policy, and further detailed review of the concept’s environmental and economic implications by key stakeholders such as Scottish Natural Heritage.
- The identification and designation of large core areas of semi-natural habitat with surrounding buffer zones, to allow the expansion and linking of existing protected areas – for example through utilisation of forest habitat networks and riparian corridors. Existing protected-area designations may be particularly suitable for this purpose.
- The development of a long-term strategy for the full rewilding of these areas, including any necessary initial restoration, subsequent management activities and, eventually, species reintroductions. This strategy should combine land acquisition plans, specific management strategies, and stakeholder partnerships.
- Widespread sustainable management of deer populations and landscape-scale ecological processes which encourage natural regeneration of woodland habitats and the creation of ecotonal habitats across Scotland.
- The reintroduction and protection (through stakeholder partnerships) of viable populations of a number of keystone species including beaver, elk, lynx, wild boar, and polecat, along with currently threatened species such as the wildcat. Programmes of protection and reintroduction can begin immediately in many
cases, with certain species (e.g. wolf and bear) requiring longer-term plans involving large-scale habitat restoration.

- Programmes of education and interpretation by a range of stakeholders to present the rationale and benefits of rewilding and reintroductions to the widest possible audience.

References


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Scottish Natural Heritage (2011a) *Scotland’s Natural Capital Asset (NCA) Index* (Inverness: Scottish Natural Heritage).


