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Literature review: Welsh upland land management practises: past, present and future

Robin Bhattacharyya-Dickson
Miriam McDonald
Robert Owen

11/06/2023

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Introduction

The British uplands are a semi natural environment, created and maintained by humans for millennia through the practises of burning, grazing and drainage, and summarily comprised of a mosaic of different habitats that support diverse faunal assemblages. The uplands are further valued by humans for tourism, recreational, livestock and game purposes, alongside having a deep cultural connection with those communities that historically have held stewardship over them. This literature review will explore the current ecology of Welsh uplands, how this links to past and present management practises, and what innovative strategies are now being used across the UK that could better serve Welsh upland areas.

Ecology of the Welsh Uplands

Uplands are typically identified by altitudinal differences between land areas. Other definitions look at the environmental conditions - higher land areas are generally cooler, with higher rainfall and poorer soils which makes them sub optimal for plant growth (Averis et al., 2004). When looking at Britain in particular, uplands are usually areas of land above which arable farming can occur (Ratcliffe & Thompson, 1988). The areas themselves can vary substantially in both geology and topography which has consequences for the habitats and species assemblages present (Fielding & Haworth, 1999), and the use and management determined from these factors further shapes their ecology. Harder, more weather resistant rock types result in steeper and more rugged mountains which has implications for both vegetation types and the fauna that use it. Many boreal and arctic bird species, for example, prefer nesting on shallower gradients, making Snowdonia and other ranges comprised of harder rock types unsuitable (Ratcliffe, 1977, Haworth and Thompson, 1990). Furthermore, these ranges are also likely to be calcium deficient leading to acidic soils that support low abundances of flora (Fielding & Haworth, 1999). It is these features that have dictated human involvement and the consequent management practises that have in turn also shaped the uplands.

Understanding the current ecological state of upland areas necessitates examining the trends in management, and consequent ecological changes, over the last few thousand years. Historically much of the Welsh uplands, at least since the last glaciation – around 10000 years ago -would've been covered in a mosaic of broadleaved forests, peatlands, bogs and dwarf shrub heath habitats. Looking at the pollen record, elm (*Ulmus procera*), field maple (*Acer campestre*), birch (*Betula pendula*) and oak (*Quercus robur*) were all common across the Welsh uplands until around 6000 years ago, after which a succession of tree species outcompeted previous stalwarts as ranges spread (Atherden, 1992). Progressive deforestation until the late 19th century then reduced this forest cover by around 90% (Bunce, Wood & Smart, 2008), with the primary drivers being construction, grazing, and burning for grouse hunting moors (Atherden 1992). Alongside these primary drivers, numerous other lesser management practises existed such as clearances to deny outlaws and vagabonds safe harbour (Linnard, 1982). These successive changes in land use and management led to an Upland environment covered with large tracts of grassland, scrub and heath land, maintained by a wide variety of grazers, from horse, cattle and sheep to wild grazers like rabbits. These tracts were interspersed with both conifer plantations and deciduous woodland, alongside wetland areas of peat bog and marshland. These areas of heathland, grassland and scrub were composed of a variety of flora that supported invertebrates, birds and small mammals, whilst the abundant peat bogs, aside from hosting their own unique species assemblages, acted as water stores and environmental balancers, sequestering carbon and protecting against drought. Changes in farming subsidies after World War 2 led to another shift in management, as sheep grazing increased to the detriment of cattle and horse grazing, which precipitated further ecological change (Ratcliffe & Thompson, 1988). Sheep farming has continued to increase and currently nearly 50% of the uplands are now used for sheep grazing (Brown et al., 2016). Abandonment is as much a problem as overgrazing with invasive species and dominating grass species spreading relatively unchecked when not subjected to some grazing pressure, a theory known as the intermediate grazing optimisation model (Grime, 1973).

Uplands today have very little native forest and wetland areas left, with large tracts given over to improved grasslands and conifer plantations (Smith et al., 2012). In fact, the typically lowland habitats of improved and neutral grassland are now more abundant than the peatlands and dwarf shrub heath habitats synonymous with uplands. The heather moorland (*Calluna vulgaris*) that remains is under increasing pressure, with 70% at risk of degradation or complete loss, and is a habitat that has high conservation value with 19 distinct plant communities, 5 of which are virtually confined to the UK (Evans et al., 2006). Calcareous grasslands that traditionally covered much of the uplands have similarly undergone large declines, mostly due to either intensification or abandonment. Conifer plantations are now more wide-spread than deciduous woodland, yet support far fewer species, whilst bog and all other types of wetland now only make up 4% of upland Wales (Bunce, Wood and Smart, 2018 – see figure 1). The loss and degradation of these already rare habitats, and the associated reduction in plant species richness, has been accompanied by widespread declines amongst the communities reliant upon them (Poschlod & Wallis de vries, 2002). Invertebrate species have seen precipitous drops in diversity and abundance as over grazed or abandoned grasslands lost floristic heterogeneity reducing niche availability (Lyons et al., 2017, Erhardt & Thomas, 1991). Bird populations, heavily influenced by seed availability and invertebrate abundances, have also witnessed large declines. Henderson et al., 2004 found that out of 35 passerine species studied 12 showed significant decreases in abundance in calcareous grassland habitats, when compared to surveys conducted between 1968-1980, which was strongly correlated with the intensification of sheep grazing over the same timeframe. Wader populations have likewise fallen with lapwings (*Vanellus vanellus*), golden plover (*Pluvialis apricaria*), dunlin (*Calidris alpina*) and curlew (*Numenius arquata*) all showing marked decreases as wetland areas were lost due to drainage (Sim et al., 2005).

Great Britain	%	Area ('000s ha)	England	%	Area ('000s ha)	Scotland	%	Area ('000s ha)	Wales	%	Area ('000s ha)
Bog	25 %	2061.6	Acid grassland	23 %	349.4	Bog	33 %	1888.1	Improved grassland	26 %	262.7
Acid grassland	17 %	1442.0	Dwarf shrub heath	18 %	269.7	Acid grassland	16 %	900.9	Acid grassland	19 %	191.7
Dwarf shrub heath	15 %	1207.7	Neutral grassland	16 %	237.1	Dwarf shrub heath	14 %	825.8	Dwarf shrub heath	11 %	112.3
Coniferous woodland	11 %	949.1	Improved grassland	15 %	224.9	Coniferous woodland	14 %	774.6	Neutral grassland	10 %	104.2
Improved grassland	10 %	814.7	Bog	9 %	133.9	Improved grassland	6 %	327.1	Coniferous woodland	10 %	102.4
Neutral grassland	6 %	525.0	Coniferous woodland	5 %	72.1	Neutral grassland	3%	183.8	Broadleaved, mixed and yew woodland	9 %	89.6
Broadleaved, mixed and yew woodland	3 %	250.1	Bracken	4 %	56.7	Fen, marsh and swamp	3 %	167.0	Bog	4 %	39.6
Fen, marsh and swamp	3 %	239.1	Fen, marsh and swamp	3 %	50.4	Broadleaved, mixed and yew woodland	2 %	120.2	Bracken	3 %	33.1
Bracken	2 %	198.4	Broadleaved, mixed and yew woodland	3 %	40.4	Bracken	2 %	108.6	Fen, marsh and swamp	2 %	21.7
Arable and horticultural	1 %	116.3	Arable and horticultural	2 %	34.1	Standing water and canals	1 %	81.9	Standing water and canals	1 %	14.8

Figure 1 - Top ten broad habitats of the UK Biodiversity Action Plan in upland landscapes in Britain. Ranked by extent in each country.

Historical and current management of Welsh uplands

The Welsh uplands have been managed using a variety of methods for thousands of years and to explore every practise in this review would be both unnecessary and prohibitive. This review will only focus on livestock management, fire management, cutting management and water management.

Historic Livestock management

Animal husbandry first appeared in Britain around 6000 years ago, with fossil evidence showing a marked increase in domesticated cattle and pig remains amongst neolithic settlements from 3800 BCE onwards (Armit et al., 2003). Goat and sheep were later added to the neolithic livestock deck around 5000 years ago, as landscapes opened up and moorlands were created from prior deforestation and cattle farming (Ryder 1964). The practise of transhumance, the seasonal moving of livestock to different grazing grounds, was widespread. Celtic pastoralists in Wales, for example, would take their milk bearing cattle and ewes to the uplands in summer to graze (Ernle, 1961). Until the 1700's livestock densities gradually increased alongside the human population, with mixed herds of cattle, horse, sheep and goat commonplace. This extensive system, with no grazing for large periods, allowed regeneration of grazed land and resulted in a heterogeneous landscape and high biodiversity. It was only from the early 1800s onwards that grazing pressure significantly increased with an increase in cattle and sheep farming, firstly across Scotland and then in Wales and other upland areas (Cunningham & Groves, 1985). The methods changed too, with transhumance replaced by fixed herds of sheep that were trained to stay in one location, minimising the need for shepherding or fencing (Hunter, 1964). This increase in stocking density, alongside the more sedentary nature of their grazing, resulted in widespread declines in both heather moorland and calcareous grassland (Stevenson & Thompson, 1993). The capability to synthesise nitrogen fertilisers from the late 19th century further exacerbated this decline as the increase in available nutrients, alongside gaps in the heather cover caused by overgrazing, allowed faster growing grasses to colonise the grazing pastures which prevented moorland regeneration (Alonso & Hartley, 1998). Grazing densities continued to increase until the late 19th century when a high point in sheep farming was reached (Cunningham & Groves, 1985).

The beginning of the 20th century brought new pressures on upland agriculture as international trade increased and the market was flooded with foreign products such as merino wool and lamb meat from New Zealand. This resulted in a steady decline among hill sheep farming which, aside from during WW1 when domestic production became a necessity, showed steadily poorer economic benefits (Cunningham & Groves, 1985, Attwood & Evans, 1961). These declines resulted in some of the first subsidies for agriculture, with a grant for drainage in 1921 being introduced, followed by grants for lime and slag in the 1930's. The advent of WW2 provided further incentives towards intensification, such as the hill ewe act in 1941, as the populace were pushed to optimise production to meet higher demand. This push set the scene for post war agriculture, and in 1946 the hill farming act was passed followed 5 years later by the livestock rearing act. Over the next 30 years the number of sheep grazed more than doubled, and grazing using other species reduced significantly, resulting in a more intensive mono-grazing system (Ratcliffe & Thompson, 1988). Supplementary feed, often paid for through subsidies, allowed higher stocking densities that further impacted the already dwindling heather moorlands (Evans & Felton, 1987), with 6% being lost to bracken and improved grassland in Wales between 1947 and 1980 (Armstrong, 1991). These moorlands often also had reduced heather cover and were in poorer condition, with 43% of Welsh moorlands reduced to <25% heather cover and 38% showing signs of damage and overgrazing (Bardgett, Marsden & Howard, 1995).

Current livestock management practises

The reformation of the common agricultural policy (CAP) in 2003 resulted in another shift in upland farm management practises, as the payments provided were no longer dependent upon production quotas but on meeting nature and heritage requirements. In Wales, further schemes were created: the Tir Gofal and Tir Cynnal agri-environment schemes in 2007, which in 2014 were combined into one scheme, Glastir, that aims to mitigate the impact of climate change whilst prioritising water management and biodiversity (Morris, Henley & Dowell, 2017). Today, more than half of agricultural land use in Wales is for grazing and hill and upland farms predominate. The most common systems in the hill and upland farms are either wholly sheep, or mainly sheep with some beef suckler cows, which together, make up 29% of farms across Wales and around 60% of Upland farms (farming facts and figures, Wales 2016). Differences between small and large holdings also exist with small holdings generally producing more diverse farm and non-farm outputs (Armstrong, 2016, Dwyer, 2018). Both these systems are generally more extensive in nature, as the land value is low, and intensification brings diminishing returns. However certain techniques are utilised to improve production, often at a detriment to the environment, and the agri-environment schemes mentioned above aim to move farms away from such practises. However, these schemes have had a variable impact in the uplands and a mixed picture has emerged due to their implementation. Generally, farming practises in the upland, being extensive in nature, have had little to no change (Arnott, 2021). When looking at certain localities however, some responses to these schemes can be identified. In Bala and Powys, for example, there has been a decrease in sheep and cattle farms of around 15-20%, above the overall drop in farms of 10%. Furthermore, practises once part of a mixed system have returned, with poultry farming increasing by around 3%. Finally, there has been a concurrent drop in stocking densities amongst the remaining sheep and cattle farms suggesting increased extensification (Lenormand, Dwyer & Devienne, 2022).

A novel method of livestock management that is growing increasingly popular involves the use of precision livestock farming technologies such as GPS collars, unmanned aerial vehicles (UAV's) and virtual fencing. These encompass the application of either single technologies or multiple tools to create an integrated system that can monitor in real time livestock on both the herd and individual level. These technologies can also impart greater control and can work remotely – a massive benefit in Upland systems where grazing pastures can be remote and difficult to access (Handcock et al., 2009). Virtual fencing, for example, uses audio cues followed by an electric shock to build associative behaviours that serve to prevent livestock from entering certain areas (Aquilani et al., 2021). This can benefit not only the livestock but also the surrounding habitats that may have a high ecological value. Other technologies allow the real time monitoring of a herd's environmental impact, allowing more informed management to reduce these negative effects. With advances in biomaterial and engineering research these technologies will become cheaper and more effective, allowing a greater uptake and increasing these technologies efficacy (Neethirajan et al., 2017).

Fire management

The earliest evidence for burning as a management tool in Britain, comes from Dartmoor at around 6000 BCE when late Mesolithic communities used fire to clear woodland (Yallop, Clutterbuck & Thacker, 2009). This was inferred through analysis of microscopic charcoal in the peat layer, the distribution of which suggested burning on a scale far larger than purely domestic fires (Caseldine, 1999). Later, neolithic peoples seemingly used burning to encourage the growth of grasses and browse which in turn would attract browsers such as red deer and cattle, increasing hunting success (Simmons & Innes, 1996, Fyfe et al., 2003). Pollen studies at Bluewater beck head, for example, showed repeated disturbance and regeneration of vegetation seemingly caused by burning (Innes, Blackford & Simmons, 2010). Cut and burn farming later followed (see below), with fire being used to clear areas of woodland to provide spaces to cultivate crops. Every few years new areas would be cleared creating a mosaic of woodland, scrub and dwarf shrub heath habitats (Simmons & Innes, 1996). These dwarf shrub heath habitats, or moorlands, created after the initial clearings, were sporadically burnt throughout the Middle Ages to improve grazing - a practise called swaling which has continued until the present day (Rackham, 1986). The burning of moorlands allows the growth of a fresh flush that provides livestock with more nutrient dense and available food.

Burning for red grouse (*Lagopus lagopus scotica*) habitat has, over the last 200 years, become the main reason for controlled burns and provides a mosaic structure comprised of old and new heather that grouse use to both nest in and feed upon (Simmons, 2003, Miller, 1980).The impact of these burns on local wildlife is complex, however, and can have markedly varying impacts depending on their frequency, extent and intensity. Burning in moderation can lead to greater heterogeneity and, consequently, a higher biodiversity but can locally disadvantage fire sensitive species (Worrall et al., 2010). If done sensibly and with respect to appropriate environments – i.e avoiding fire sensitive habitats like blanket bog - a moderate level of burning can thus be beneficial. Golden plover, hen harriers (*Circus cyaneus*), curlew and merlin (*Falco columbarius*) all benefit from moorlands under appropriate burn management whilst invertebrates likewise are present in greater abundance and diversity, and are food for a host of other species (Yallop Clutterbuck and Thacker, 2010). If no burning occurs, or occurs very infrequently, heather will eventually enter the degenerate phase, where active growth of new shoots and flowers decline, the canopy opens out, and ground cover decreases (Gimingham, 1972). This degenerate phase will last for around 10 – 15 years after which the heather will die, and other pioneer species may colonise (Schellenberg & Bermeier, 2022). If burnt in this degenerate phase, regrowth will be extremely slow, with a loss in diversity and persistence of bare ground for many years after the burn (Hobbs & Gimingham, 1984). Intense burning, or frequent burning, can also lead to a loss of biodiversity as the heather either has too little time to regenerate, or cannot regenerate, as all propagative material has been burnt (Hobbs & Gimingham, 1984). Moorland subjected to too severe a burning regime generally progresses to acid grassland – a habitat that supports far fewer species and lower abundances (Hobbs & Gimingham, 1984).

Whilst burning on heathland has been practised for centuries and, under the moderate regimes discussed and when on appropriate sites can have benefits for a variety of bird and invertebrate species, there has been a change in the intensity of burn management regimes in recent years (Yallop et al., 2006b). Since the 1970's, burn frequency (the time between repeated burnings of the same area) has increased across English uplands and bogs, with the median time between burns decreasing from 19.2 years to 15.8 years (Yallop et al., 2006). Anecdotally there also seems to have been an increase in the number of "hot burns" - where fires are driven against the wind to increase the temperature or are set in the height of summer – which greatly influences what vegetation regrows (Yallop et al., 2006). Hotter fires result in a near monoculture of heather and when combined with grazing regimes this effect is exacerbated. Which areas of land are managed using burning is another contentious issue and burning of bog and on Sites of special scientific interest (SSSI) is all too frequent with 32% of SSSI's in poor condition because of burning regimes (Yallop et al., 2006). It is important to note that there is limited data on the extent of burning across the UK's upland areas and trends can be localised with large regional variations. Prescribed burning can also expose the peat layer beneath, leading to increased runoff that has negative impacts upon drinking water, and an increase in the release of dissolved and particulate organic carbon (Holden et al., 2012).

This variability in fire management practises and the associated negative impacts has led to calls to develop an ecological approach to fire management, where the practise prescribed considers: vegetation structure and species composition, plant responses to fire and associated traits, fuel flammability, and the impact of both individual fires and the fire management practise as a whole (Davies et al., 2008). This could include the burning of only certain aged heather for example, as young heather not only regenerates far better than older areas but the fires produced are more manageable and require less resources to control (Davies, 2005, Davies et al., 2006). The designation of fire free zones that could be legally enforced could also be established that could prevent fires on areas that respond poorly to burning. Peatlands would benefit greatly from such a measure as although official guidance advises against the burning of peatlands no framework exists that strictly prohibits such a practise (Davies et al., 2008). This more considered approach would allow fire management in areas that would benefit from such a measure and keep cultural ties to the land intact whilst preventing it in areas that would suffer. .

Cutting management practices

Cutting as a management technique has likewise been employed in upland Wales for millennia, with patches of forest first cleared to provide space for agriculture and to provide fuel and tools for neolithic peoples. Since the advent of the first heather moorlands grazing and burning have been the primary drivers managing these habitats, but with the advent of engine powered machinery came a new ability to cut vegetation on similarly large scales. This had an influence on these existing management practices which can fall short or have detrimental effects in certain situations. Distinct from previous practices where vegetation would be cut before being burnt in order to create clearances for agriculture, this practice generally relies on mechanical flails to crop large areas of vegetation down to a desired height, with the litter created either being left or used for fodder. Often used for heathland management it achieves a similar effect to burning but through a slightly different mechanism. Whilst burning opens up canopies and stimulates seed germination, cutting stimulates bud growth from stools - similar to the regrowth experienced when coppicing (Usher, 1992). Consequently, the heathlands managed using these differing methods have different characteristics and can be environmentally beneficial, or detrimental, dependent upon various local factors and other concurrent management practises (Milligan et al., 2004). Grazing pressure is one such factor that needs to be limited for regrowth to occur. The age of the heather stand can also limit the effectiveness of the regime implemented. Old stands are both more vulnerable to invasion by bracken (*Pteridium aquilinum)* and purple moor grass (*Molinia caerulea*) and less suitable for vegetative regrowth. This decreases the effectiveness of a cutting regime and is a potential pitfall even if the heather stand was burnt (Liepert et al.,1993).

Today, cutting is being increasingly utilised as it neatly avoids the detrimental effects associated with burning. Uncontrolled fires, damage to wetlands, impacts on local fauna and exposure of the peat layer leading to water runoff issues are all either reduced or avoided completely (Sanderson et al., 2020). Additionally, the heather and associated faunal assemblages have a similar response to cutting as to burning, with heather often showing more rapid regrowth and invertebrate communities showing a similarly increased abundance and diversity in the years after the cut (Liepert et al., 1993). However, there are negatives and limitations associated with this regime. Firstly, cutting of the heathland relies on machinery that cannot access certain topographies as surfaces are too steep or rocky to drive upon. Secondly the labour involved is not only greater, but the rotation time is shorter, which has additional impacts on the cost of implementing such a regime.

Historical and current water management practises

Uplands generally have very complex hydrological systems, with a high number of watercourses, high rainfall and a variety of wetland habitats. Various methods of managing this system have been utilised within the UK for hundreds of years and has been accompanied by marked ecological changes. Two of the most widespread and impactful of management practises are drainage and river channelisation, which are connected to some degree but have distinct ecological and environmental consequences.

Peatlands are an ecosystem of both national and international significance and are areas of land with naturally occurring layers of peat that are typically formed under waterlogged conditions from decaying plant material - generally *Sphagnum* moss species (Bain et al., 2011). They are important for a variety of reasons having not only a high biodiversity value but acting as a carbon sink and water store whilst providing recreational and, increasingly, tourism opportunities (Kimmel & Mander, 2010). The UK has a large proportion of these ecosystems containing between 10-15% of peatlands within Europe, and 13% of blanket peatland worldwide which places a greater significance, and greater consequences, on how they are managed (Bain et al., 2011). Despite their global rarity and ecological value, over the past century peatlands have been drained across the UK in order to reclaim land for agriculture and game, mainly through the creation of open ditches that channel water from the waterlogged peat layer into local watercourses (Holden et al., 2004). This land drainage effort was incentivised by the introduction of a land drainage subsidy post WW2 and has been the main contributing factor behind the UK's reputation as one of the most extensively drained countries in Europe (Baldock, 1984). Throughout the 60's and 70's peatland loss approached 20000 hectares per year with an estimated 80% of peatlands now damaged or deteriorating, often drained for livestock and forestry purposes (Stewart & Lance, 1983, Littlewood et al., 2010).

Peatland drainage has a variety of impacts upon both biotic and abiotic parameters and can severely affect the interactions between vegetation, soils, aquatic ecosystems and hydrological systems. Drainage directly impacts high water table dependent species like sphagnum moss but can also result in the spread of unfavourable grasses (Coulson, Butterfield & Henderson, 1990), a contradictory effect given drainage often occurred to reclaim land for livestock. A lowered water table also has implications for the soil as air can enter more readily, affecting microbial processes and increasing decomposition (Holden, Chapman & Labadz, 2004). Severe peat shrinkage becomes increasingly likely and the consolidation of these now dry peat layers often leads to considerable subsidence. Water storage capabilities are also affected which has knock-on effects for soil moisture content and leads to gradual loss in organic matter releasing CO_2 rather than absorbing it (Tiemeyer et al., 2007). Further to this, the drains themselves are a major source of sediment as the bare walls and floor are susceptible to freeze thawing in the winter and desiccation in the summer, releasing particulates which are then washed downstream. Finally, drainage allows greater interaction between water and soil layers with a higher nutrient content than the anaerobic peat soils present pre drainage. This leads to nutrient runoff which is channelled into local watercourses and has further implications for stream biota (Holden, Chapman & Labadz, 2004).

In 1985 the government subsidies for peatland drainage ceased and in recent years an effort has been made to reverse this drainage, mainly by blocking existing drainage ditches to keep water on the land. This has frequently been successful in reducing sediment runoff, preventing erosion, facilitating revegetation and reducing flood peaks (Worrall et al., 2010) and whilst this practise has limitations, generally only being effective on smaller drains and on shallower slopes (Evans et al., 2005), the effect on soil and stream hydrology can be substantial. Price, Heathwaite & Baird, 2003 showed that the water table reached levels commensurate with intact peatland, allowing revegetation of much declined peatland species whilst Shantz & Price, 2006 found this rise in water table level increased overland water flow further benefiting wetland species. Research into the effects of drain blocking on downstream biota is scarce but can be hypothesised. The rise in water table level has been shown to reduce erosion which in turn reduces the amount of sediment washed into watercourses whilst the dams themselves can prevent what sediment is produced from entering these watercourses (Holden, Gascoign & Bosanko, 2007b). This reduction in sediment suspension and the resultant alteration of stream physiochemistry will more closely mirror that of an intact site (Ramchunder, Brown & Holden, 2009).

River channelisation is the modification of rivers for the purposes of flood control, navigation, drainage and erosion prevention, and generally involves the straightening and reinforcing of riverbanks and beds. Evidence of channelisation in Britain stretches back for 500 years but has been particularly prevalent since the 1940's, when the land drainage act made government grants available for land drainage purposes (Brookes, Gregory & Dawson, 1983). This had an impact not only on the lands being drained – mostly peatlands of global rarity and high ecological value – but also on the river ecosystems themselves and on downstream ecological communities. Habitat loss along lengths of channelised rivers was severe as the increase in flow rate, deepening of the channel and the loss of meanders, led to a loss of species adapted to slower flow rates, a loss of habitat and morphological features such as pool riffle sequences and undercut banks, and the loss of vegetation and in stream cover (Brookes, 1985, Gibson & Power, 1975). Following channelisation of the Little Sioux River in Iowa, for example, 54% of habitat was lost immediately following the river modification (Hansen, 1971). Channelisation also acted to cut watercourses off from their floodplains, which reduced the exchange of water and matter between floodplain and watercourse and often resulted in a greater risk of flooding downstream (Chartered Institute of Water and Environmental Management, 2014).

Channelisation has long been recognised as detrimental to local ecology and efforts to restore river ecosystems have been ongoing from as early as the 20th century (Roni et al., 2012). Initially focusing on enhancement structures such as flow deflectors, artificial riffles and rubble mats, river restoration now focuses on restoring river processes. One way this is achieved that has an immediate and lasting impact is to "re-meander" the watercourse by taking out bank reinforcements and riverbed structures and reintroducing bends. This re-meandering, alongside more sustainable catchment land management, the restoration of riverside forests, and riverine wetland creation, has considerable impact both on short term floral and faunal responses and on long term ecosystem services (Gilvear, Spray & Casas-Mulet, 2013). Further to this, a meta study investigating the effects of multiple river restoration projects observed positive effects across numerous indicators with macroinvertebrates, fish and other river dependent vertebrates, and floral communities all increasing in number and diversity (Kail et al., 2015). Even within agricultural catchments improvements were observed although, as noted above, the level of intensiveness of catchment land management had substantial impacts on the outcome of the project.

Another novel approach that is gaining steady traction is the reintroduction of certain species that either through direct action, or through their effect on the behaviour of other species, have positive impacts for ecosystems as a whole. Beavers (*Castor fiber*), for example, have profound impacts on the surrounding ecosystem, creating habitat for other species, providing ecosystem services, changing hydrological systems in ways that further benefit surrounding floral and faunal communities, improving water quality and raising water tables to name a few (Brazier et al., 2021). Beaver reintroductions to Scotland have so far yielded very promising results, with habitat quality improved and ecosystem services restored alongside providing socio economic benefits (Gaywood, 2018). Appropriate management is required however as negative impacts can occur to certain habitats of high conservation value.

Conclusions

Historical management practises have had a profound impact on our current landscape and have been culturally significant and had economic benefits for a large proportion of upland communities. Many of the practises, however, have associated negative impacts on both the abiotic and biotic spheres, and have become outdated and unnecessary, often serving only to maintain tradition, whilst some past practises that have been neglected in recent years may have potential applications that have been overlooked. New approaches are required that serve both the environment and the people and communities that rely on the uplands for cultural and economic gain. More research into such approaches needs to be undertaken to establish effective implementation and to discover new methods, or rediscover old methods, that can benefit both the Uplands and the communities living there.

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