**The Carbon and Water Guidelines**

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|  | **Windfarm on peat** |  |

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**Chapter 1: Introduction**

*1.1 Overview of ‘The Carbon and Water Guidelines’*

The purpose of this document is to synthesise, for the non-expert, knowledge of the processes that control carbon (C) export from the terrestrial environment to catchment drainage systems. Understanding this is particularly important for those whose activities may increase loss of C from the terrestrial environment and for those who have to monitor, manage and mitigate such losses. The increasing interest in retaining C in terrestrial stores from multiple agencies has resulted in several ‘best practice’ documents (further outlined in chapter 5), in lieu of legally enforceable directions. In this document we synthesise these different approaches to provide clearer guidance on shaping activity to minimise increases in carbon losses via aquatic systems and provide guidance on methods to quantify aqueous carbon losses from field sites (chapter 7).

To complement this focus, we outline the current legislation pertinent to drainage system export of carbon and consider the degree to which environmental impact assessments consider these exports as part of the development process (chapter 2). We provide an overview of the processes that produce mobile forms of organic carbon within catchments and that govern the export of carbon via catchment drainage systems (chapter 3), we discuss the fate of the exported carbon (chapter 4) and we summarise future environmental challenges in understanding and managing aquatic C export (chapter 6). This introductory chapter provides a summary overview of key points which are further expanded in the subsequent chapters.

*1.2 The rationale for the ‘The Carbon and Water Guidelines’*

There are three key reasons for monitoring aquatic carbon export from catchments. The growing scientific and societal interest in these reasons underpins the need for clear supporting information for non-experts. This information should not only provide guidance for practitioners but also offer scientifically sound and up-to date supporting information that can be consulted to aid the development of further initiatives. The three supporting reasons are:

1. *We need to know if export of C via drainage waters reflects changes in the amount of C that is retained in terrestrial C stores.* There is a growing appreciation of the size, importance and significance of the terrestrial soil carbon (C) pool (e.g., Dawson & Smith 2007, Ostle *et al.* 2009) such that with a projected changing climate we seek to reduce atmospheric CO2 emissions and preserve C losses from carbon landscapes (biomes rich in carbon). Of particular conservation interest are peatland systems which represent significant terrestrial carbon stores which have accumulated slowly. The relative ease of quantifying drainage system dissolved organic C concentrations over prolonged periods, has revealed that the concentration of dissolved organic C has been increasing in surface waters across most of Europe and North America in response to reduced acid deposition on land surfaces (Monteith *et al.* 2007). It is unknown if this also reflects reduced C sequestration in soils.
2. *We need to know if land-use change has influenced the carbon balance within the catchment.* Human disturbance of carbon-rich soils can change how carbon is retained within the terrestrial system. Organic carbon losses from soils in the form of dissolved (DOC) and particulate (POC) forms are most easily observed and quantified by monitoring the drainage system (e.g., Worrall *et al.* 2004, Worral *et al.* 2007) and such monitoring programmes can reveal disturbance-driven changes in C exported via drainage systems due to increased contribution from disturbed terrestrial sources (e.g., Tetzlaff *et al.* 2007, Grieve & Gilvear 2008/9, Waldron *et al.* 2009, Waddington *et al.* 2010). Catchment monitoring programmes are effective as they integrate spatial exports from the whole catchment. If the sampling points are chosen wisely aquatic monitoring presents fewer challenges than quantifying gaseous losses for the same area, where tall towers and eddy-covariance systems are needed to achieve integration at a sufficiently large spatial scale (e.g., Lafleur *et al.* 2003).
3. *Increases in dissolved and particulate organic C export can impact on the receiving catchment drainage* *waters*. Losses of terrestrial carbon can impact on water quality. Organic anions make a significant contribution to bog drainage waters (Hruska *et al*. 1996) and humic substances in solution are linked to complexation and mobilisation of metals such as aluminium and iron (Tipping *et al*. 2002), thus increased C export to receiving waters may also reflect an increase in metal export (e.g., Tetzlaff *et al*. 2007). The role of DOC in contributing to colour is yet to be fully elucidated (e.g., Holden 2009) but more strongly coloured water usually reflects increases in DOC concentration. As society prefers clear water, increases in the colour of upland waters have consequent implications for drinking water treatment costs and effectiveness (Watts *et al*. 2000). Additionally increased aqueous carbon exports can impact on ecological systems: DOC and POC contain biologically active compounds which support microbial activity in soils, streams and lake (e.g., Jones *et al*. 1998, Clair *et al*. 1999, Cole *et al*. 2002; Hopkins & Gregorich, 2004) and higher DOC and POC concentrations can reduce light penetration in streams and shallow lake systems (V-Balogh *et al*. 2009).

Given the knowledge that developments within carbon-rich landscapes can increase C losses from the soils and increase the C load in the aquatic drainage system, guidelines, similar to those developed for the management of forests and their associated hydrology, will be useful for stakeholders with responsibilities for managing these landscapes. Stakeholders here include developers, planners and the regulatory agencies with responsibility for environmental protection. Further, with carbon footprinting becoming more commonplace, in conjunction with the need to define an ecologically acceptable carbon load in aquatic systems, future legislation may require C export from a landscape to the drainage system to be quantified. Thus it is timely that knowledge associated with this subject is collated.

The UK government Natural Environmental Research Council ([www.nerc.ac.uk](http://www.nerc.ac.uk)) has funded a knowledge exchange network, CLAD (Carbon Landscapes and Drainage; [www.clad.ac.uk](http://www.clad.ac.uk)) to facilitate interaction between the research community and stakeholders to better understand process and issues associated with terrestrial-aquatic C transfer. For example a key discussion topic has been the expansion of renewable energy developments on peatlands and to what extent this impacts on catchment drainage systems. Such discussions both through and outwith CLAD have helped shape the structure and content of this document. However, the Carbon and Water Guidelines presented here are designed to be relevant in a context broader than renewable energy developments, and offer information that supports any aspect of how changes to terrestrial C stores may impact on aquatic C export.

*1.3 Peatlands and carbon landscapes.*

In this document we use the term ‘carbon landscape’ to refer specifically to landscapes which can sequester significant amounts of carbon and thus have the capacity to act as globally important carbon sinks. Examples include peatlands, forested areas and coral reefs. The ‘Carbon & Water Guidelines’ focus on terrestrial carbon landscapes and the integrated flux to the drainage system. We will use the term carbon landscape where the discussion is more broad-reaching than peatlands alone, or when multiple local definitions for a landscape component exist, or when the catchment, although not peat, is a driver of C export to catchment drainage. However, in the Northern hemisphere, the carbon landscapes which provide the greatest terrestrial carbon storage are underlain by organic (peat) and organo-mineral soils and thus much of our focus is on their export of C to drainage systems.

Peat soils have developed at suitable sites in the Northern Hemisphere (e.g., Steinmann and Shotyk 1997, Smith *et al.* 2004) since the retreat of the polar ice sheets at the end of the last ice age, between 16 000 and 8 000 years ago (the timing of this varies latitudinally). Peat develops at sites which are poorly-drained and where the water table sits at, or close to, the ground surface. The primary peat-building plants are *Sphagnum* moss species which partially decompose as they die. However, decomposition slows down in the anoxic layer below the water table such that losses of organic material are greater than gains, allowing the partially decomposed organic material to accumulate incrementally.

The definition of peat varies among different national soil classification systems and also among different disciplines (and so we may substitute with the term carbon landscape). For example, in the UK, the Soil Survey of Scotland (1984) defines peat as ‘the organic layer or layers exceeding 50 cm depth from the soil surface and with an organic matter content of greater than 60 %’. The UK Forestry Commission (Cannell *et al.* 1993) used 45 cm as the critical depth. The soil classification for England and Wales defines peat soils as an organic soil layer more than 40 cm deep, or if overlying bedrock more than 30 cm deep, with a minimum organic carbon content of 12–18 % (Avery, 1980). The Canadian Sphagnum Peat Moss Association defines peatland as ‘A specific type of wetland on which extensive organic material has accumulated. These areas with peat-forming vegetation growing on peat have an undrained layer of peat at least 12-18 inches (30-45 cm) deep’2. These differences in definition have the potential to cause some confusion when comparing between countries. Further in cool humid temperate areas such as Scotland or Scandinavia, large areas are covered by organo-mineral soils (soils such as peaty podzols and peaty gleys which have a surface organic layer less than 50 cm deep). For the field scientist and land manager identification of peatlands may be problematic where organo-mineral soils and peat can frequently be found in a mosaic within the landscape. Disturbance of either peat or organo-mineral soils can exacerbate C exports and we therefore use the generic term ‘carbon landscape’.

*1.4 Global significance of carbon landscapes*

*1.4.1 C stored in the landscape is C not in the atmosphere*

We focus here on peatlands as guidelines already exist for afforested areas and their catchment management1 and peatlands are the globally important terrestrial C store accessible by man and therefore susceptible to land-use change. Peat soils contain more carbon per unit area than any other habitat on Earth (Joosten & Couwenberg 2008). Northern peatlands represent about 90 % of global total peatland C pool of 612 Gt C and more than 90 % of global peatland net carbon balance (Zicheng 2011). Such soils cover approximately 15 % of the UK land area. The most recent estimates of UK peatland carbon storage are of 2302 Mt C, with approximately 70 % of this in Scotland (Fig. 1.1; Billet *et al.* 2010, Table 1.1).

**Table 1.1** Estimates of UK peatland carbon stocks (adapted from Billett *et al.*, 2010)

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Country** |  | **Soil depth** | | |
|  | **Area (km2)** | **0–100 cm**  **(Mt C)** | **>100 cm**  **(Mt C)** | **Total**  **(Mt C)** |
| Scotland | 17789 | 1104a | 516a | 1620 |
| England | 4246 | 296b | 123d | 419 |
| Wales | 732 | 67b | 52c | 119 |
| Northern Ireland | 1873 | 90b d | 54 | 144 |
| UKs | 24640 | 1557 | 745 | 2302 |
| aChapman *et al.* (2009); bBradley *et al.* (2005); cSmith *et al.* (2007); dPro rata to Scottish stocks below 1 m | | | | |

When peatland is disturbed, some of the peat may dewater. Stored carbon which is thus exposed (via several routes) to air is readily converted to CO2 and released to the atmosphere. The size of the store is such that conversion of just 5% of the UK’s peat would release the same amount of CO2 to the atmosphere as the current annual CO2 production of the UK economy. Thus the UK carbon budget is sensitive to relatively small changes in the quantity of peat C storage. Peatland carbon is not only vulnerable to direct human disturbance and changes in management (e.g., Ward *et al.* 2007, Armstrong *et al.* 2010), but also to climate change (e.g., Dise 2009, Clark *et al.* 2010a, Clark *et al.* 2010b).

*1.4.2 Records of past environment help contextualise modern environments*

Peatlands are also important due to the information that can about their depositional environment e.g., environmentally-induced historical changes in peat and carbon accumulation rates (e.g., Belyea & Malmer 2004; Mäkila & Moisanen 2007; Beilman *et al.* 2009) and environmental changes such as increased acid deposition (e.g., Malmer & Wallen 2004).

For example accumulation rates over the last 150 years at four UK sites ranged from 35.1 - 209.1 g C m2 yr-1 (Billet *et al.* 2010). Greatest accumulation rates generally occurred in the acrotelm (less decay of recently added organic matter) and in raised mire sites (Butterburn Flow in Northumbria, UK and Laxford Bridge, Scotland, UK) with rates greater than 100 g C m2 yr-1 since the mid-20th century (Billett *et al*. 2010). The raised mires are wetter sites dominated by *Sphagnum*, considered to be the key peat-forming species. The lowest rates were found at Lochnagar (NE Scotland, UK), which has relatively low primary productivity and is a sloping blanket mire site dominated by vascular plants rather than peat-forming *Sphagnum* and with a relatively dry surface due to more rapid runoff. Comparison with paleo-accumulation rates showed inter-site differences in C accumulation rates slowing (Lochnagar) or not changed significantly (Laxford Bridge) since the 1970s.



**Figure 1.1** Peat and peaty soils of the United Kingdom (map reproduced from JNCC 2011). Deep peat soils (dark brown), shallow peaty soils (green), wasted deep peat soils (light brown). Peat in South-East England is largely fen peat. Reproduction by permission of OS on behalf of HMSO@ Crown copyright and database Right 2010, MLURI 100019294, AFBI 1:50000 soil digital Data, National soil Maps @ Cranfield University, BGS 1:50000digital data (license 2006/072)

Inter and intra-site differences are likely to be influenced by differences in vegetation and hydrology, which can occur over multiple scales and so paleoenvironmental analysis can reveal changes in C accumulation rates in response to key environmental contols such as hydrological regime or temperature, although as our knowledge expands, unpicking these becomes more complex. Thus for example the capacity to reconstruct past climate wetness from paleo-water tables is being challenged (Swindles *et al*. 2012). Reconstruction of dominant past vegetation matrices are possible from soil macrofossil analysis (e.g., Belyea & Malmer, 2004) however reconstruction of past aquatic C export is not known to be possible. Thus paleoenvironmental analysis does not yet have a role in constructing past aquatic C losses. Rather it can underpin modern day analyses of how much C terrestrial systems must sequester and transfer to soils for storage.

Modern accumulation rates are calculated from C budget assessments of net ecosystem exchange and aquatic losses. For continued C sequestration, aquatic carbon export represents the minimum sequestration rate in catchment soils in order for there to be no net loss of this terrestrial store. Thus knowledge of past C accumulation rates are important to contextualise whether there has been a change in the capacity of modern day systems to accumulate C and how is controlled by local and regional environments. Further information on accurate calculation of aquatic C export budgets can be found in chapter 7.7

*1.5 Development and resource use in peatlands*

Societies have always sought to use available resources although over time the ranking of what was valuable of sites site has changed. Although there is currently increasing focus on the value of peatlands as a long-term global C store, in the past priority uses were different. For example, in the years following 1945 in the UK a systematic programme to drain large areas of peatland was implemented with the aim to increase the productivity of the UK uplands for sheep grazing (Holden *et al.* 2007). A considerable area of UK peatland has also been afforested, particularly since the formation of the Forestry Commission in 1919 and between the 1950s and 1980s (Turner & Mackey 1995). There has also been a substantial increase in the area of peatland managed by burning for grouse populations (Yallop & Clutterbuck 2009). It is now recognised that such development activities that were historically practised at large scales on peatlands may have impacted on peatland form and functioning, and thus on C exports via catchment drainage systems. Relevant processes affected include loss of carbon through biologically-mediated oxidation as the water table is lowered for drainage (Freeman *et al.* 2001) and oxidation during burning resulting in loss of carbon via a number of hydrological pathways (Ward *et al.* 2007, Yallop & Clutterbuck 2009).

More recently there has been an increase in a number of other developments, the impacts of which are less well understood. Probably the most important of these is a large increase in the number of wind farm developments sited on peatlands. These are complex developments with a number of construction subcomponents which have the potential to increase carbon losses both directly to the atmosphere and via the drainage system (Grieve & Gilvear 2008/9, Waldron *et al.* 2009, Murray, 2010) including direct removal of peat to accommodate turbine bases and laying of power cables, felling of forestry located on peat and drainage of the peat near roads and other infrastructure (Nayak *et al.* 2010).

Other activities occurring on peat which may impact on the functioning of peatlands and thus change aquatic C export include the removal of peat overburden for mining developments, road building (as part of national road building programmes or on private land), construction, extraction of peat for horticulture and fuel, infrastructure projects, burning for heather management and also potentially infrastructure associated with new hydropower developments. Whilst some of these activities do not involve the removal of peat from site (and so are not considered extractive processes for regulatory purposes), they often involve extraction, compression, and translocation of peat within the site (sometimes unintentionally) and therefore are likely to disrupt the hydrological connectivity and ecological functioning of both the peatland ecosystem and the translocated peat. Such disruption has potentially major implications for biodiversity, water supply and the functioning of peatlands as carbon sinks (Lindsay 2010).

The impact of developments on carbon processing and sequestration, hydrological functioning and biodiversity of the carbon landscape is still the subject of on-going research, but catchment drainage studies have been valuable in confirming that there have been measurable impacts in some cases (e.g., Tetzlaff *et al.* 2007; Grieve & Gilvear 2008/9).

*1.6 Which form of C export is most important?*

This is discussed in more detail in chapter 3, but summarised here. Losses of carbon to surface waters draining carbon landscapes occur through three major pathways: fluxes of carbon dioxide (CO2) from decomposition of carbon compounds dissolved or suspended in drainage waters; fluxes of dissolved organic carbon (DOC) arising from dissolution of carbon compounds from organic soil layers in the water draining through the soil; and fluxes of particulate organic carbon (POC) usually eroded from organic soils or soil layers by flowing water. Which form of C dominates aquatic export from carbon landscapes is not easily answered as this is likely to vary depending on the location and timing of sampling. However, in landscapes without significant peat degradation, DOC losses are generally an order of magnitude larger than losses in other forms and POC losses are generally slightly larger than losses in gaseous form (e.g., Dawson *et al.*, 2002; Alvarez-Cobelas *et al.*. 2012). These losses are subject to large inter-annual variability and there can be significant regional variability. For example, POC losses from the heavily disturbed carbon landscapes of the southern Pennines in England (Table 1.2) are probably more significant both in relative and absolute terms than for other landscapes due to the greater influence of erosional processes. Here “peat block” erosion and transport following undercutting of river banks contributes to POC (Evans & Warburton, 2001). Thus if resources are limited we would advocate focus on quantifying DOC over other pools.

**Table 1.2** Estimates of POC export per unit area (from multiple sources, referenced)

|  |  |  |
| --- | --- | --- |
| **Country & WaterBody /Catchment (ha)** | **C-flux**  **(kg C ha−1 yr−1)** | **Source of estimate** |
| ***Scotland*** |  |  |
| Brocky Burn — upper (0.68) | 9.0 – 21 | Dawson *et al.*. (2004) |
| Brocky Burn — middle (0.83) | 8.2 – 19 | Dawson *et al.*. (2004) |
| Brocky Burn — lower (1.3) | 5.9 – 28 | Dawson *et al.*. (2004) |
| Water of Charr (14.2) | 20 – 175 | Dawson *et al.*. (2004) |
| Small Burn (0.41) | 4.6 – 12 | Dawson *et al.*. (2004) |
| Burn of Waterhead (3.4) | 4.4 – 16 | Dawson *et al.*. (2004) |
| Water of Dye — upper (24.6) | 5.4 – 19 | Dawson *et al.*. (2004) |
| Water of Dye — lower (46.3) | 5.9 – 26 | Dawson *et al.*. (2004) |
| River Don, Parkhill (1273) | 5.3 ± 4.0 | Hope *et al.*. (1997a) |
| River Dee, Park Bridge (1844) | 1.9 ± 1.2 | Hope *et al.*. (1997a) |
|  |  |  |
| ***Northern England*** |  |  |
| N. Pennine, Moor House (11.4) | 27 – 317 | Worrall *et al.*. (2003a) |
| N. Yorkshire Moors (9.0–300) | 2.3 – 5.0 | Arnett (1978) |
|  |  |  |
| ***Mid-Wales*** |  |  |
| Upper Hafren (0.93)±95% C.I. | 27 ± 19 | Dawson *et al.*. (2002) |
| Afon Cyff (0.04) | 8.8 | Reynolds (1986) |
|  |  |  |
| ***Canada*** |  |  |
| Canagagigue creek | 0.5 | Dance *et al.*. (1979) |
|  |  |  |
| ***Russia*** |  |  |
| Don | 1.53 | Kempe (1985) |
| North Dvina | 7.28 | Kempe (1985) |
|  |  |  |
| ***Sweden*** |  |  |
| Ore | 4.08 | Ivarsson & Jansson (1994) |

*1.7 Summary*

This chapter has introduced the role that carbon landscapes play in carbon storage. Carbon landscapes also provide other vital ecosystem services such as protection of aquatic systems, supply of drinking water and contributing to the maintenance of biodiversity. However, Carbon landscapes are vulnerable to a range of pressures from such as burning heather to encourage regrowth for grouse habitat, direct removal for horticultural peat, disturbance for development and some agricultural practices. In view of their importance and vulnerability, we outline in this document current process-based understanding of how carbon is lost from the landscape to the drainage system and simplify for the stakeholder the multiple guidance documents that exist on minimising this. Multiple guidances exists in lieu of legislative principles, but legislation may come in response to increasing emphasis on quantifying the C cycle and understanding how it responds to natural and anthropogenic drivers. It is timely then for ‘The Carbon and Water Guidelines’ to be the first non-academic document that summarises research to increase understanding of processes and provides original practical guidance (e.g., for estimating aquatic C exports ) or synthesises existing guidance in a single document.

**Chapter 2: Legislation of relevance to carbon landscapes and water**

Although there are many initiatives which indirectly offer some form of protection to carbon-rich soils and landscapes, at present there is no single piece of legislation specific to the protection of peat or soil carbon and the C budgets of their drainage systems. Furthermore, the carbon concentration in surface waters is not a determinand that has to be monitored in drinking waters, although colour is. As we move towards routine carbon footprint determination or become further concerned about levels of C in surface waters (for the reasons outlined in section 4), monitoring C levels and calculating export budgets may become a legal requirement. In this context we outline here a number of pieces of more general environmental legislation that have relevance to peat and soil carbon and could prime development of C-specific legislation. Where relevant we identify components of the legislation that may extend to carbon landscape losses to drainage water. We outline first international directives, before considering national and local legislation and voluntary codes. Our national-level examples are primarily from the UK but we recognise that similar codes exist for other countries.

*2.1. The Kyoto Protocol (1997)3*

This protocol to the United Nations Framework Convention on Climate Change (UNFCCC) has the aim of "stabilising greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system”. Land use activities resulting in emissions are included under Articles 3.3 and 3.4. At the 2010 UNFCCC meeting in Copenhagen, draft rules were agreed to enable peatland and rewetting of drained areas to be included in national emissions accounting. Further information about this can be found in the IUCN ‘Kyoto Protocol and National Accounting for Peatlands’ briefing document4. It seems logical that if such rewetting is to be included its viability in reducing soil carbon losses, ultimately to reduce atmospheric emissions, will have to be validated and here quantification of catchment drainage losses post rewetting will be necessary.

*2.2. Water Framework Directive (WFD)5*

This over-arching piece of European legislation has been designed to consider impacts on the aquatic environment and protect the ecological status of surface waters, groundwater and coastal waters.  Ecological status is based on hydromorphological, ecological and physic-chemical parameters. It does not however, consider aquatic forms of carbon as a physico-chemical parameter although it does recommend DOC monitoring.  Carbon in water, by affecting water colour, transparency and pH can affect the ecology of aquatic environments. As such dissolved organic carbon is used in the UK lake typology which underpins how ecological status is defined in lakes. This suggests it should be more prominent as a water quality variable with WFD. DOC concentrations can affect other water quality parameters.  Removal of water colour and fulvic and humic acids is also a significant cost in providing potable water (Dawson *et al.*. 2009). As such the assertion here is consideration should be given to that DOC being included as a key physico-chemical variable.

Suspended sediment is a physico-chemical variable under the WFD and thus incorporates POC. Direct measurement of POC would make sense in terms of its ecological significance in aquatic environments and the carbon loss it represents from catchment soils. Thresholds on suspended solids loadings to protect the aquatic environment may trigger some actions to reduce soil erosion into rivers.

The WFD also offers prescribed limits for nutrients and if future legislation was aimed at controlling DOC export to reduced aqueous CO2 emissions (from respiration, section 4.1) then legislation that controls macronutrient levels (P, N) is important. Carbon and macronutrients can co-vary in drainage waters (e.g., Waldron *et al.*. 2009) and macronutrient availability can influence whether C is sequestered in aquatic biomass or quickly respired. Thus statutory nutrient monitoring components of the WFD could prime estimates for appropriate drainage catchment C levels.

*2.3. EU Drinking Water Directive (DWD)Council Directive 6*

While carbon content is not a primary focus of this legislation there is a requirement to monitor for total organic carbon . Monitoring can take place in the treatment works or in the supply zone if it can be demonstrated that there is an adverse effect on the parameter under consideration (although no monitoring protocol is prescribed). The directive also demands that colour is monitored. Components of colour are often used as a proxy for dissolved organic C concentrations (e.g., Worral *et al.* 2006, Tipping *et al.*, 2009) and there will therefore be information on components of catchment C export. Here colour is of concern due to the consequences for human health during water purification and thus limitations on colour could prime legislation on monitoring C export in drainage.

*2.4. EU Common Agricultural Policy (CAP) payments7*

Pillar I of CAP makes payments to owners of agricultural land on an area basis as long as their activities meet Good Agricultural and Environmental Conditions (GAEC). At present these do not include maintaining and preserving soil carbon stocks or controlling C inputs to local drainage, although plans to reform GAEC requirements might allow scope for this. These payments only apply to agricultural land, not sporting estates or afforested land. Pillar II of CAP makes payment for participation in various agri-environment schemes and in Scotland this is administered by Scottish Rural Development Plan (SRDP). Presently, the incentive for this payment is not targeted at enhancing soil carbon stores or reducing drainage C exports, but if increased C stocks in soils are considered of benefit to agriculture, it could be a target. Uptake of Pillar II schemes is low.

*2.5. The UK Climate Change Act (2008)8*

This Act does not include much consideration of soil carbon, but within the nation’s emissions budget (covered by emissions reduction targets) does include changes in soil greenhouse gas (GHG) fluxes (including CO2 and CH4) as a result of land-use and land-use change. However this does not specifically protect soil because it does not assess changes in biomass inputs to soil. Also the land uses considered within the Act are very broad (such as arable, grassland, forest), and fluxes are averaged for each of these across the UK, although they may have more detail on soil type/temperature and rainfall regimes. Thus the outcome is not very sensitive to site-specific management of e.g., the drainage or peatland management of interests to readers of these guidelines.

*2.6. The Climate Change (Scotland) Act (2009)9*

This Act, passed by the devolved Scottish Parliament, is similar to the UK Climate Change Act in that it includes soil C in national emissions budgets, but not aquatic C emissions. Although the Act uses emissions factors which are more suited to Scottish landscapes, it still does not allow detailed consideration of land management influences. However there are some additional provisions which are pertinent to soil carbon, in particular the requirement in Section 57 to produce a Land Use Strategy for Scotland by March 2011 (now achieved) which aims to provide some strategic planning with regard to land use. However, it is not yet clear how far this will go towards protection of soil carbon. There is also a power in Section 58 to vary the times of muirburn (discussed further in section 2.7). Finally Section 44 imposes duties on public bodies to contribute to emissions reduction including those resulting from decision making. This reduction emission has not yet been applied to aquatic systems e.g., if land management changes result in a greater aquatic DOC pool that subsequently is respired to CO2 and so efflux increases, then section 44 may be relevant to monitoring C in drainage systems.

*2.7. Muirburn Code (2011)10*

Muirburn is the practice of burning old heathland vegetation to encourage new growth for grouse feeding The Muirburn Code gives statutory guidance on how heather burning should be carried out, with the primary aim of protecting nesting birds and property, rather than preserving soil C. Practitioners believe that there may be scope to update the Code to protect soil C, but at present there is some question regarding is the definition of best practice and how to enforce it. Continued discussion of section 58 in the Scotland Climate Change Act (2009) is relevant here as the intention of this clause was originally to protect ground-nesting birds if the timing of the nesting season changed. However, such flexibility might allow better timing of muirburn to avoid damage to underlying peat e.g., after a particularly dry spell when a deeper burn could occur (ref), or to minimise losses to the drainage system e.g., by disallowing muirburn prior to likely wetter periods.

*2.8. Scottish Soil Framework (2009)11*

This document aims to protect soil C and was developed with advice from a wide range of stakeholders. It is aimed at Scottish Government policy leads, delivery partners, environmental and business NGOs, research organisations and other key stakeholders with an interest in soils. It describes key pressures on soils, particularly climate change and relevant policies to combat those threats, and identifies the future focus for soil protection, key soil outcomes and actions across a range of sectors. However, it does not introduce any new policy or legislation which might have binding requirements for those developing or managing peatlands. It could be that monitoring of catchment export would be required to assess if changes in land-use had resulted in increased lateral export to the drainage system.

*2.9. UK Forestry Commission Standard12*

The UK Forestry Standard (UKFS) is the reference standard for sustainable forest management in the UK. The UKFS is underpinned by a series of Guidelines, describes the context for forestry in the UK, outlines approach of the UK governments to sustainable forest management, defines standards and requirements and provides a basis for regulation and monitoring.

The Forests and Water Guidelines1 suggest that actions are taken to ‘Ensure the removal of forest products from the site, including non-timber products, does not deplete site fertility or soil carbon over the long term and maintains the site potential.’ This allows interpretations resulting actions focussing on drainage pathways.

The Forestry and Climate Change Guidelines13 recognise that peat can represent a larger carbon store than forests and recommends that Forestry on peat deeper than 50 cm is avoided and the impacts of disturbance associated with forestry are considered (although carbon losses via aquatic pathways are not specified). It is also recommended that planting next to bog habitats is avoided.

*2.10. Formal requirements for considering C loss to drainage in an environmental impact assessment.*

There is no UK legislative requirement that carbon emissions or management be included in the planning process assessments of a development on peatland carbon landscapes. Thus there is no formal requirement of the environmental impact analysis (EIA) that an analysis of the impact of any development on peat includes an assessment of the potential impacts on carbon sequestration, losses and gains. However in 2008 the Scottish government first commissioned and published its ‘Carbon Payback Calculator for Wind Farms on Peatland’14, subsequently revised and republished in 2011. This is an open access spreadsheet-based tool which calculates the period of time which a windfarm development on peatland will take to pay back the carbon lost to the atmosphere as a result of its construction. In the revised calculator, carbon losses include those associated with the construction processes, from land use change such as forest clearance, from drainage and from disturbance of peat soils (Nayak *et al.* 2012).

Increasingly renewable energy developments on peatlands are subjected to an analysis using the Carbon Calculator for Windfarms on Peatland. The insistence by statutory consultees that developers or their representatives undertake an analysis using this tool represents an interesting development in the move towards carbon accounting and management in peatland carbon landscapes and their drainage systems; the calculator has become a *de-facto* requirement of the planning and consenting process without a formal legislative requirement for its use. However there are no statutory limits on payback times for these developments and consent appears to be granted following dialogue between the developers and the consultees.

The requirement for windfarm developments to undergo assessments with the carbon calculator is not shared with other types of development which may have similar or greater impacts on peatlands yet provide no carbon payback during their operational lifetime (e.g., roads, housing, and opencast mining). To some extent the move towards carbon accounting for peatland development and management may be viewed as an evolutionary process and the development of the Carbon Payback Calculator may be a focal point stimulating further developments in this area. Some consultants use the payback calculator as a tool to determine aspects of carbon budgeting for renewables developments not sited on peat. This can be helpful in informing management of the development process to reduce carbon emissions and represents an interesting wider application for the tool perhaps not originally envisaged by its creators. The structure of the calculator is such that each component offers viable process-specific calculations (with some linkage to development generic variables e.g., size of site) and thus an approach similar to this, offers potential in estimating a footprint of any landscape development that changes C export to the drainage waters. This need to calculate the carbon payback through such an adapted application could be incorporated into future legislation that requires estimates of carbon export to drainage to be managed.

**Chapter 3: Production, transfers and export of aqueous carbon within and from catchments**

Drainage routes in carbon-rich landscapes are the key pathways for the transfer of terrestrial carbon from peat soils to lakes, oceans and the atmosphere. To understand aqueous carbon losses and the potential for their manipulation by human activities it is important to understand first the mechanism of aquatic C production with these landscapes and secondly the different hydrological pathways through which water passes through terrestrial C stores and thus can export C. In this chapter we first summarise the processes by which DOC (generally the largest aqueous C pool) is produced in organic soils and soil horizons and then we discuss the hydrological controls of aqueous C export. Finally we outline of the production and export processes of particulate organic carbon and dissolved inorganic carbon.

*3.1 Formation of dissolved organic carbon (DOC).*

There are thought to be two broad types of DOC. The first DOC type is produced by the roots of living plants and subsequently modified by bacteria (Kuzyakov, 2002). Much of this material is produced in the growing season and quickly lost from the system (within a year). Export of this material is thought to contribute to the marked seasonal profile of higher [DOC] in temperate drainage systems in the late summer months observed in time series datasets (Fig. 3.1 below). This DOC pool represents rapid C cycling within the peatland C.

The second form of DOC lost via drainage is believed to be formed during anaerobic fermentation processes. In aerobic soils, including peat layers above the water table, decomposition of organic matter is mediated by microbes in the presence of oxygen, ultimately converting complex carbon molecules to CO2 . This CO2 is lost to the atmosphere, or flushed away during rainfall or by water movements with the soils as the soils become wetter (as dissolved inorganic carbon, section 3.9). In waterlogged soils such as peat, decomposition can also follow a rather more complex pathway (Fenner & Freeman 2011) and is indirectly retarded by the consumption of oxygen that occurs below the water table. The breakdown of organic matter in these environments does not progress to CO2 due to the presence of a group of chemicals known as phenols (Freeman *et al.* 2001). Phenols are produced when plant structural components such as lignin are broken down (Elder & Kelly 1994). In the presence of oxygen, phenols are further broken down by microbial enzymes called phenol oxidases. In the absence of oxygen (consumed during decomposition in the almost static water table), phenol oxidases are not produced, allowing phenol to accumulate, dissolved in the water (Freeman *et al.* 2001, 2004). These phenols represent a significant proportion of DOC and prevent the action of other enzymes needed to complete the decomposition process (Wetzel 1992). Thus, DOC is produced as part of a partial degradation process occurring under anaerobic condition. Many of these enzymes are extra-cellular and therefore DOC produced is in the pore water and not the microbial cell.

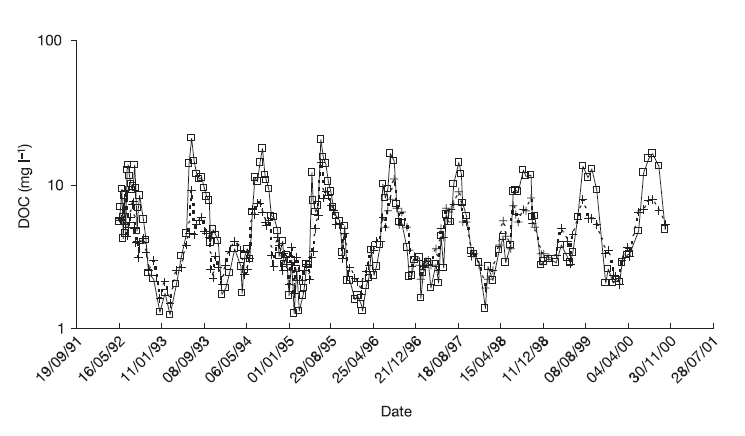


Fig 3.1. Marked seasonal changes in [DOC] from two catchments in Wales (Freeman *et al.* 2004).

Given this understanding of decomposition in peatlands, it becomes clear that production of DOC is indirectly related to movements of the water table. When the water table drops, new areas of the acrotelm become exposed to oxygen and aerobic decomposition occurs with DOC as an intermediary product, but CO2 as the primary end product (Fenner & Freeman 2001). When the water table rises and the area of the acrotelm now below the water table becomes anaerobic, relatively more DOC is produced. This DOC is flushed periodically as DOC-rich water is replaced with new water from rainfall events.

Changes in the water table are linked to the hydrological budget of the catchment and this can often be strongly seasonal (although current norms are projected to be disrupted by changing climate, Trenberth *et al.* 2007). To help understand how these controls are manifest in catchment C loads we first consider hydrological controls on DOC export and then the seasonality that exists in this export.

*3.2 DOC in the different flow pathways of peat and organo-mineral soils*

Hydrological transport and the effect of changing flow pathways on DOC concentrations in organo-mineral soils are relatively well understood. DOC concentrations increase with stream discharge because at large discharges proportionally more flow is generated from the upper organic horizons and litter where DOC is mainly produced, than from the mineral horizons (Hope *et al.* 1994, Schiff *et al.*.1997 Hinton *et al.* 1998). Under low flow conditions (small discharge values) the largest proportion of flow is derived from mineral subsoils where DOC concentrations are small, and indeed can be further reduced by sorption on iron and aluminium (oxy)hydroxide minerals.

Historically a number of scientists have described a peatland as a ‘sponge’, slowly releasing water in dry periods, and reducing flooding by absorbing heavy rainfall. This has been challenged (Holden & Burt 2003) following observations that water tables generally remain relatively high but baseflow may not be maintained, and flow regimes in peatland catchments are generally rather flashy, responding very quickly to rainfall. Thus flow pathways in peat are best understood within the context of the acrotelm-catotelm model (e.g., Holden & Burt 2003). This model differentiates faster flow in the upper ‘active’ peat layer which has higher hydraulic conductivity and a fluctuating water table (acrotelm), from slower flow in the more ‘inert’ lower layer which has slower hydraulic conductivity despite being permanently saturated (catotelm) due to the reduction in permeability as a function of peat compression and decomposition increasing density. The catotelm is generally considered to be anoxic, whereas the acrotelm is oxic and therefore it is within the acrotelm that DOC production is more important. The thickness of the acrotelm is strongly influenced by the position of the water table (e.g., Holden & Burt 2003).

Due to their capacity to transmit water macropores and pipes are hydrologically important in peatlands. Definitions of macropores vary widely (Luxmoore 1981, Watson & Luxmoore 1986, Baird 1997) and generally include other spaces found in surface areas of the acrotelm such as those created by invertebrates e.g., worms, and cracking caused by desiccation. Macroporosity decreases by two orders of magnitude over the first 20 cm depth of peat but is greater in *Sphagnum*-dominated areas. Under saturated conditions, macropore pathways are significant for water flow in the upper layers of peat (uppermost 5 cm) (Holden *et al.* 2005).

Peat pipes (see Fig. 3.3) are an intriguing aspect of peatland hydrology. Pipes can be found within a few centimetres of the surface to depths of three meters, with diameters ranging from 3-70 cm and can be over 100 m in length (Holden & Burt 2002). Hydrological response times to event flow are relatively short but pipes continue to discharge for some hours after a rainfall event, suggesting that they can contribute disproportionately to base flow (Holden & Burt 2002). Pipe flow was calculated to contribute approximately 10% of stream volume but up to 30% of flow during rainfall events (Holden *et al.* 2001). Pipes can infill and therefore may not be permanent; similarly new pipes can form (Holden *et al.* 2011).

Clearly the depth at which water interacts with the peat and the duration of this interaction will influence how C-rich the drainage waters are. Using five hypothesised routes for rapid hydrological responses in peat-dominated catchments (Evans *et al.* 1999) we can consider how these flow pathways influence C export.

1. *Infiltration-excess overland flow:* water input is greater than the rate at which water can infiltrate into the peat surface. This type of flow is more likely to happen with a compacted, frozen or hydrophobic surface and we may expect run-off from the peatland surface to be low in DOC.



Fig 3.2. Overland (Photo P Chapman)



Fig 3.3. Pipe flow (Photo P Chapman)

1. *Saturation-excess overland flow*: soil is saturated resulting in surface ponding and flow (See Fig. 3.2). This type of flow is more common in temperate systems and as there is relatively short contact time with the soil pores we may expect DOC concentrations to be low.
2. *Rapid flow within the acrotelm* (*or percolation excess*) caused by ponding at the acrotelm/catotelm interface. Where DOC is available for transport, this flow will likely be high in DOC.
3. *Rapid flow at the acrotelm/catotelm* interface despite a lack of saturation in the catotelm. Where DOC is available for transport, this flow will likely be high in DOC.
4. *Pipe flow*: water movement through naturally-occurring pipes below the surface of the peat (Fig. 3.3). This source of flow will be generally high in [DOC] due to the intimate contact with the peat body, although at high flow rates [DOC] may be reduced due to reduced transit time.

During rainfall events there will be changes in which flow pathways dominate delivery of C to the drainage systems. Once the peatland surface is no longer hydrophobic (e.g., Worral *et al*. 2006b) initial losses to drainage are driven by percolation excess or rapid through-flow at the catotelm-acrotelm interface. As the event progresses and the water table rises there is an increasing tendency towards saturation excess and with water moving over the peat surface (Worrall *et al.*, 2002).

Radiocarbon studies of peatland-derived DOC in aquatic drainage have characterised DOC to be predominantly very young (less than 5 years old) (Benner *et al.* 2004, Tipping *et al.* 2010, Billet *et al.* 2012); this DOC therefore represents a loss of relatively recently sequestered carbon. Pipes also seem to export DOC of the same age (i.e. young carbon) as that found in overland flow (Billet *et al.* 2012) but can remove ‘old’ carbon as particulates, indicating that their pathway within the peat is erosive but they channel flow from the surface, where the DOC pool contributes more to export.

New hydrological pathways can be created during droughts as cracks and fractures form in the peat and create routes which bypass the acrotelm - here DOC is not flushed in the ‘normal’ manner (Evans *et al.* 1999) and so DOC concentrations in catchment drainage may change. However, spatial differences in runoff can occur suggesting there may be preferential flow paths within the peat (Holden & Burt; 2003). Changes to hydrological pathway can result in increased runoff and thus increased DOC flux e.g., from wetland areas in Sweden during the 1970s and 1980s (Forsberg, 1992). However, the long term increase in DOC concentrations does not always mean that there has been a change in hydrological pathway (e.g.,, Worrall & Burt, 2004).

*3.3 Seasonal variations in DOC*

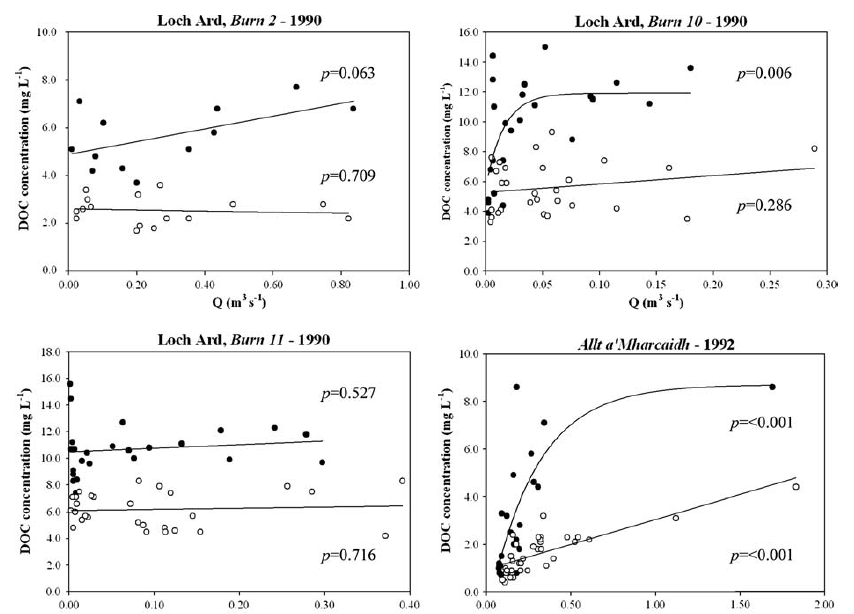
The finding that most DOC exported from peat is young carbon is likely related to the strong seasonal cycle of DOC production and export. In temperate zones this is typically manifest as small concentrations of carbon in stream and river water during the winter months before increasing steadily from the initiation of the growing season as temperatures increase and the water table generally is drawn down (ref) reaching a maximum in the early autumn, before a steady decline back to the winter minima (Fig. 3.4).



Fig 3.4. [DOC] at differing peat depths in the Pennine Hills, England, (Clarke *et al.* 2008), reveals that the seasonal changes in [DOC] are most apparent in the shallow depths, reflecting the interaction of seasonal production and hydrological export and upper layers more susceptible to temperature change and with faster hydrological connectivity to stream flow.

Where seasonality exists in DOC concentrations, aqueous carbon losses can only be accurately quantified if the sampling programme covers an appropriate temporal scale to capture seasonal changes, best defined as a complete hydrological year. If significant periods of the year are not sampled, DOC export estimates will be underestimated if maximum [DOC] is not captured and overestimated if minimum [DOC] is not captured. Accommodating the seasonality may allow more accurate construction of DOC export from the relationship with discharge. This is demonstrated well in the analysis of long-term monitoring data from six sites across Scotland (Fig. 3.5; Dawson *et al.*, 2008). For each site the relationship with discharge is improved by categorising [DOC] by season. There are also site-specific differences in the discharge-DOC relationships and we return to this study in consideration of spatial variation (section 3.5 below).

In areas subject to winter freezing such as Canada and Scandinavia, maximum DOC concentrations are often found during the snowmelt flow peaks during late spring and early summer. Here DOC remaining in soil pore waters after the previous autumn together with that generated by freeze-thaw actions on plant and microbial material is flushed by the melting snow and ice (e.g.,, Finlay *et al.* 2006, Ågren *et al.*. 2010).



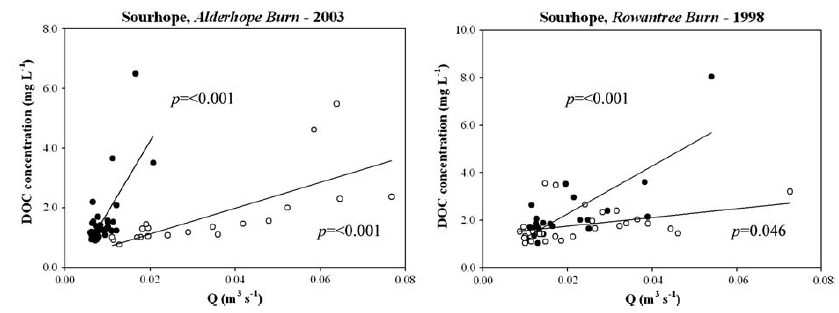
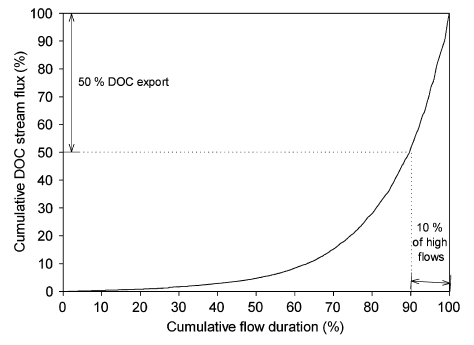


Fig 3.5. DOC-discharge relationships for sites across a rainfall gradient in Scotland (taken from Dawson *et al.* 2008). Dark circles summer-autumn data. Open circles winter-spring data. Note differing scales.

*3.4 Changes in DOC in response to changing flow*

DOC concentrations have been found to increase significantly with discharge for a number of peat-dominated sites (Tipping *et al.*. 1988; Grieve 1994, Hope *et al.*. 1997, Dawson *et al.*. 2008) although some studies have shown that this relationship may not be prevalent throughout the year (Dawson *et al.* 2002, Clark *et al.* 2007, Dawson *et al.*. 2008).



**Fig 3.6.** The importance of high flows in controlling carbon loss from peatlands (from Clark *et al.* 2007). This indicates that the highest 10% of flows are responsible for 50% of the DOC export through the interaction of generally higher concentrations in larger volumes of water.

Rainfall events have a particularly important influence on the amount of DOC transferred from carbon landscapes to drainage systems. In organo-mineral soils a single large rainfall event can be responsible for more than 50 % of the annual carbon losses from catchment soils (Hope *et al.*, 1997) and this seems to also apply to peat-dominated catchments (Clark *et al.* 2007, Fig. 3.6). Observations that event flow is responsible for the majority of C losses indicates clearly the importance of event sampling when determining C losses and mechanisms to accommodate this sampling are discussed further in Chapter 7.

*3.5 Spatial variations in DOC concentrations*

There are two aspects to consider here: within soil DOC production variation and within drainage export variation. Significant variation has been observed in [DOC] at different depths of the peat (Clarke *et al.* 2008) (Fig. 3.4) although these differences are further overprinted by seasonal controls.

Spatial variation in DOC concentrations within a catchment has been observed, particularly at low flow. This variation has both internal and external controls. The internal controls are discussed further in Chapter 4 but if there is no further significant inflow of DOC depending on the rate of C reprocessing there may be downstream reduction in DOC concentration. In this section we this focus on external controls:

External controls include catchment characteristics and local climate as these influence landscape production and hydrological balance. DOC concentrations have been shown to be strongly related to the percentage of peat or carbon-rich soils within a catchment (Hope *et al.* 1997, Aitkenhead *et al.* 1999, Kortelainen *et al.*, 2006) which in turn will be influenced by slope such that distinctive soil types have both differences in DOC production potential and transit time of through-flow. In the UK, the latter can be classified as the Hydrology of Soil Types (HOST) classes and this has proved useful in identifying areas of the catchment that are important in generating flow (e.g.,, Tetzlaff *et al.*, 2007b)

Land use will be also related to the area of C-rich soils in a catchment and it is possible for adjacent catchments with different land uses to have quite different [DOC] concentrations even though climatic conditions and underlying geology may be similar (Fig. 3.7). Similarly spatial variation in DOC concentrations may occur reflecting intra- or inter-catchment variation in land disturbance caused as disruption to the land-surface allows increased oxidation or organic matter and changes hydrological flow pathways (e.g.,, Tetzlaff *et al.*. 2007, Waldron *et al.*, 2009).



A

B

Fig 3.7: [DOC] (panel A) and [POC] (panel B) in Whitelee N- and S-draining catchment waters. Each sample point is the mean ± 1 SD of the four N- or five S-draining catchments sampled on the same day. Higher [DOC] is found in the S-draining catchments which contain more peatland whilst the N-draining catchments have more improved pasture.

The six catchments previously considered for their seasonal variation in export (Fig. 3.5) also showed spatial differences in [DOC] interpreted to be influenced by a climate gradient from western to eastern Scotland (Dawson *et al.*, 2008). The three wetter Loch Ard sites did not exhibit a distinct autumnal maximum, but continual production and relatively rapid export as the higher precipitation in the summer/autumn months continually removal of DOC. The three eastern sites (Allt a’Mharcaidh and Sourhope) with a characteristic autumn DOC flush were drier and had lower rainfall–runoff ratios, longer transit times and annual drying and wetting regimes.

Studies such as this (Dawson *et al.*. 2008) indicate that inter-site comparisons of aquatic DOC losses estimated from drainage budgets must assess first if there are natural regional differences. To what extent there is spatial variation is a scale issue and thus must also be considered in monitoring and sampling programmes (Chapter 7).

Despite inter-site differences in DOC concentrations there are wider scale geographic controls on surface water DOC concentration changes. Increases in stream and lake DOC concentrations in Northern Europe and North America over the last 30 years (Fig. 3.8) are considered strongly related to reductions in atmospheric deposition of sulphate caused by a reduction in coal burning and therefore SO2 emissions to the atmosphere (Monteith *et al.* 2007). At the smaller scale this regional control can be mediated by further drivers of change in DOC losses such as changes in nitrogen deposition, land management practices such as drainage and burning (Clarke *et al.* 2010).

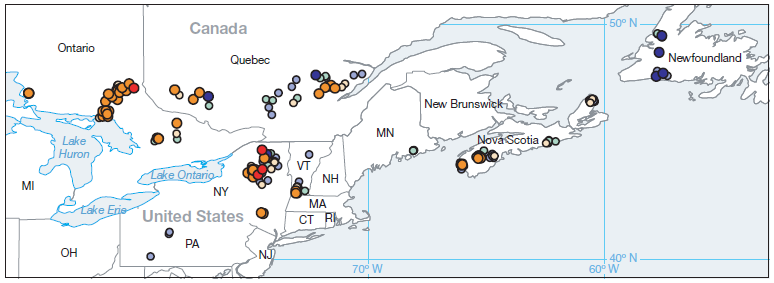
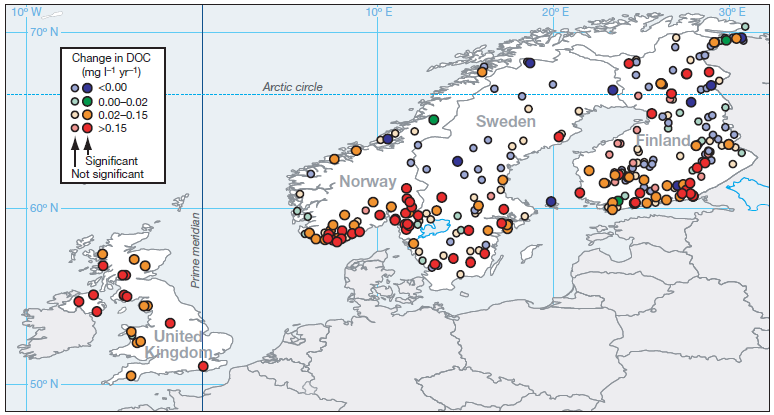


Fig 3.8. Trends in DOC (mg l-1 yr-1). Data for monitoring sites on acid-sensitive terrain in Europe (above) and North America (below) 1990–2004. The circles size and colour represents changes in [DOC] and whether significant (large circles) or not (small circles). From Monteith *et al.* 2007.

*3.6 Processes leading to the formation of POC*

Particulate organic carbon in streams consists of fine particles of peat (organic sediment) eroded from peat surfaces, physically degraded terrestrial vegetation, and material generated within the stream. Loss of particulate organic carbon (POC) from peatlands via aquatic pathways is an understudied area. While a number of studies (Hope *et al.* 1997, Tipping *et al.* 1997, Dawson *et al.* 2002, Billet *et al.* 2010) have found that DOC is the single largest form of carbon lost from carbon landscapes via drainage, POC losses have been shown to be larger in some instances (Pawson *et al.* 2008, Billett *et al.* 2010).

Land management activities and drainage can increase POC production. Non-storm POC concentrations increased in response to the percentage of the catchment area used as arable land (Hope *et al.*. 1997). Track use can generate POC (Robroek *et al.* 2010). The production of POC has been found to be significantly greater from drained peats than in undrained peats where erosion occurs (Mayfield & Pearson, 1972), and peat drains can be an important sources of sediment (Holden *et al.*. 2007). Carling & Newborn (2007 – in Holden 2009) found sediment yields were up to three orders of magnitude greater in drains relative to streams.

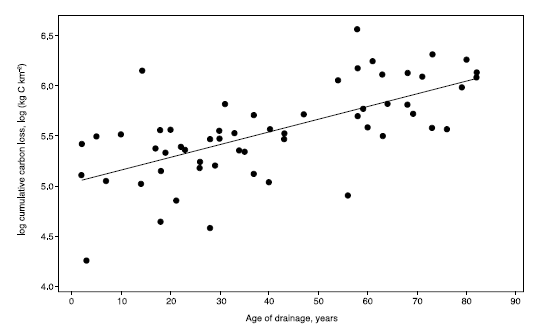


Fig 3.9. Loss of POC vs. age of drainage showing that the longer the pipe is in existence the more significant a role it has in POC export (from Holden 2006)

However, the most important control of POC in peat drainage systems is the extent to which eroding sediments are linked to the channel – (the slope-channel linkage concept, Evans &Warburton, 2005).) Examination of sediment loads in an eroding peatland found that the main source of sediment was erosion of drainage channel floor and edges (Evans and Warburton, 2005). Some of this material is present as eroded peat blocks which are produced due to erosion and undercutting of the channel edge. These features have recently been shown to contribute significantly to the sediment (and therefore POC) budget of the system (Warburton & Evans 2011). In very heavily eroded and unvegetated systems slope channel linkage can be very efficient resulting in considerable losses of sediment to drainage channels (Evans *et al.* 2006).Without-efficient slope-channel linkage eroded material is unlikely to be transported to drainage. Thus it seems reasonable to suggest that physical disturbance on the edges of drainage systems might increase drainage POC and a geomorphological assessment of prior may be an important activity to inform the design a monitoring programme.

Peat pipes may also be a significant source of POC losses increase as the age of the pipe drainage increases (See Fig. 3.9 above). This is probably due to aging processes which occur within the pipe including erosion, and the drying and freeze-thaw which creates more friable erodible peat for export by the physical action of flowing water (Holden *et al.* 2011). The age of this POC exported in pipes in English Pennines peatlands varied between modern and 653 years BP, suggesting that both newly-fixed and old carbon are being lost from the peat via this route (Billet *et al.* 2012).

*3.7 Relationships between POC and flow*

Positive relationships between POC and flow have been observed for non-eroded peat-rich catchments in Scotland (Hope *et al.*, 1997). Similarly, a logarithmic relationship between POC and stream discharge (See Fig. 3.10 below) was observed in the eroded English Pennine peatlands (Pawson *et al.*, 2008). This suggests that most POC is lost via drainage during storm events with concentrations ten or 100 times greater at high flow compared to low flow. This increase in [POC] during the course of an event is simply demonstrated by changes in the amount of material retained by glass fibre filters (Fig. 3.11 below).

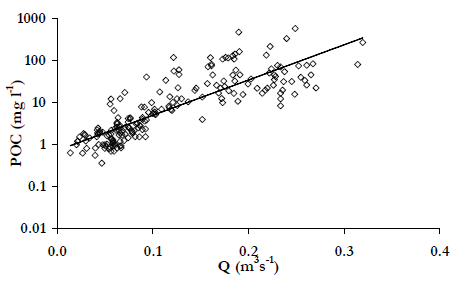


Fig 3.10: POC concentrations in stream waters in the English Pennines increase with discharge (Q) Note the logarithmic POC scale indicating the increase in [POC] with discharge is not linear but higher flow export proportionally more C than lower flows (From Pawson *et al.* 2008).

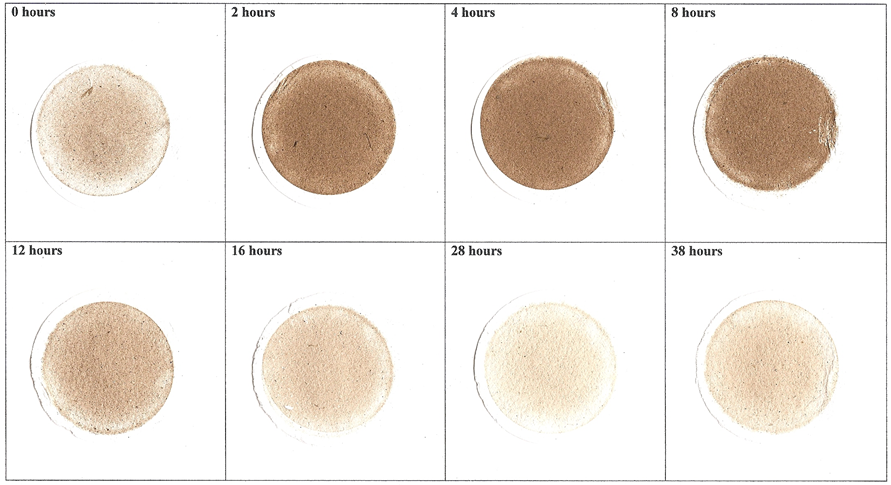


Fig 3.11 POC losses from a stream over the course of rainfall event. Samples have been collected on glassfibre filters. (Photo courtesy of I. Grieve)

*3.8 Spatial variation in POC losses*

As documented above, land management practices such as drainage will vary spatially giving rise to spatial variation in POC losses. For example, differences in vegetation cover could be responsible for sedimentary losses an order of magnitude greater in heavily eroded unvegetated sites at a site in the South Pennines relative to sites revegetated following erosion in the North Pennines (Evans *et al.*, 2008). Differences in POC losses between similar paired sites in Scotland were interpreted by as being due to differences in physical disturbances by livestock and humans (Dawson *et al.*, 2008).

More complex intra- and inter-catchment differences in POC were revealed by Hope *et al.* (1997) who mapped out differences in POC losses within two Scottish catchments. Strong positive correlations of POC concentrations with the percentage of peat cover within sub-catchments indicated that peat was a major source of POC. Interestingly, POC concentrations released during non-storm flow appeared to be related to the extent of arable land within the sub-catchments (Hope *et al.* 1997). This observation raises the issue of ensuring that only peatland POC is considered during monitoring.

*3.9 The role of dissolved inorganic carbon (DIC)*

In addition to the losses of organic carbon in the form of POC and DOC, streams also transport dissolved gasses such as CO2 (which is derived from soil respiration) and carbonate ions (HCO3- , CO32-), which are derived from underlying rocks and hydration of CO2 in water under conditions of increasing pH. Collectively CO2, HCO3-and CO32- are known as dissolved inorganic carbon (DIC) (e.g.,, Talling, 2010).

CO2 concentrations in streams are primarily driven by inputs of groundwater and soil pore-water and are a function of the strength of the connection between the land and the stream i.e. the volume and CO2 concentration of terrestrial waters entering the stream and the volume and surface water CO2 concentration already present (Griffiths *et al.* 2007). The fate of that CO2 in the stream depends on physiochemical controls on concentration and capacity to degas; temperature, velocity, turbulence and wind speed all exert varying degrees of control on CO2 losses to the atmosphere (Rebsdorf *et al.* 1991, Dawson *et al.* 2001a). One particularly important control is the establishment and activity of biotic communities particularly biofilms which grow on stream surfaces. These include diatoms and other plants which consume CO2 as part of photosynthesis and bacteria which consume DOC and produce dissolved CO2.

Significant diurnal shifts in concentrations of DIC have been ascribed to the activity of in-stream biological communities (e.g.,, Dawson *et al.* 2001b, Waldron *et al.* 2007, Talling 2010, Nimick 2011). Photosynthesis consumes CO2 and is light-dependent, so diurnal changes in CO2 concentrations can be significant (Guasch *et al.* 1998, Dawson *et al.* 2001a, Neal *et al.* 2004). However, this control may sometimes be obscured by overwhelmingly large inputs from terrestrial sources (Griffiths *et al.* 2007). In temperate systems there is a seasonal component to this in-stream processing of carbon (Dawson *et al.* 2011). This has been shown to be largest during the summer when lower flow allows for stable surfaces thus enabling biofilm establishment (and respiration), and increasing photosynthesis (Dawson *et al.* 2001a, Griffiths *et al.* 2007, Battin *et al.* 2008). If drainage systems have trees in the riparian zone light will be more limited and diurnal biological responses may be limited and respiration will dominate throughout the day.

Concentrations of the differing forms of DIC in a stream will change along its course in response to the particular characteristics of the catchment being examined. One study measuring the changes in CO2 and HCO3- along the course of a peatland stream in NE Scotland found that as the stream passed through peatland areas CO2 concentrations increased, suggesting that peat is a significant CO2 source (Dawson *et al.*, 2001a). A progressive loss of CO2 along the stream length was interpreted as being due to outgassing as CO2 equilibrates with the atmosphere. Outgassing has been found to be enhanced by turbulence (Rebsdorf *et al.* 1991, Billett & Moore 2008). Wiith more dependence on physical processes controlling changes in concentrations that increasing biotic controls, changes in DIC downstream have been described as following a peatland stream continuum (Dawson *et al.* 2004), considered than the river continuum concept (Vannote *et al.* 1980).

**Chapter 4: Fate of exported material**

Dissolved or suspended carbon is exported from terrestrial stores by artificial or natural drainage sytems. Organic C flux has been estimated to be between 10 and 100 kg ha-1 yr-1 for UK rivers (Hope *et al.* 1997), with the larger estimates associated with the more peaty catchments. Total UK C losses were estimated as 0.69 ± 0.07 × 1012 g-1C yr-1 for DOC and 0.20 × 1012 g−1 C yr-1 for POC (Hope *et al.* 1997), but these estimates did not include storm event data and so are unsurprisingly lower than estimates which do accommodate event export e.g.,, total organic C flux of 1.4 × 1012 g−1 C yr-1 (± 30 %, Cannell *et al.* 1999).

As soon as C enters a drainage system, in-stream processes begin to transform and cycle C. These processes continue such that during aqueous transport, carbon is lost from the water column and the net effect is usually a downstream reduction in DOC, POC and CO2 concentrations. The principal mechanisms of carbon loss are:

* Loss to the atmosphere of dissolved CO2 (present either from direct import or produced by microbial or chemical reaction with dissolved organic matter).
* Sedimentation of POC in depositional areas such as lakes and estuarine areas.

As streams and rivers flow into estuaries and the marine environment, fresh DOC inputs from the surrounding soils become relatively less important compared to inputs from upstream (Minshall *et al.* 1985, Dawson *et al.* 2001). In areas of higher population densities, pollution or where land disturbances have occurred (Armentanao & Menges 1986, Waddington & McNeill 2002) there can be large C influxes via discharge of treated and untreated sewage. In larger rivers, a combination of young and old DOC, and predominantly old POC (>1000 yr), from terrestrial sources is found, but generally the younger, more labile DOC is ‘lost’ leaving an older OM component to enter estuarine and oceanic C pools (Raymond & Bauer, 2001).

An awareness of the fate of exported material in streams is necessary to contextualise intra- and inter-catchment differences in river water C concentrations and export loads.

*4.1 Loss to the atmosphere of dissolved CO2.*

If CO2 concentrations, [CO2], in an aquatic system are over-saturated with respect to atmospheric equilibrium, CO2 can be degassed to the atmosphere. This is true for any dissolved gas.

*4.1.1 Magnitude of the loss*

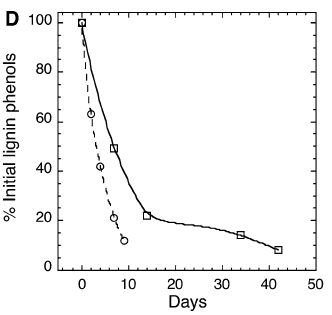
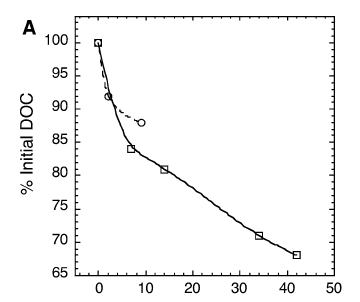
Historically, losses of CO2 (and CH4) by degassing in headwaters, rivers and wetlands (ponds and lakes) have been largely ignored in the majority of published riverine flux measurements and budgets. However, degassing is a potentially important route by which C is returned to the atmosphere from both lakes (e.g., Kling *et al.* 1991, Cole *et al.* 1994, Cole & Caraco 2001) and rivers (e.g., Billett *et al.* 2004). Considerable variations exist in gaseous C efflux data from rivers and lakes in the UK and North America, possibly due to variations in mineralisation rates and the saturation levels of the soils (Dawson & Smith 2010) but gaseous C evasion can be of the same magnitude as the total amount of C transported laterally downstream (e.g., Hope *et al.* 2001, Billett *et al.* 2004). Large intra-catchment variation exists in gaseous C losses from surface waters depending on factors such as topography, topography, soil type and hydrological flow paths (Dawson *et al.* 2001, Hope *et al.* 2001) but as with DOC regional variations can exist that reflect climatic variation. For example, a study of CO2 efflux from streams and rivers in the US found regional variation in CO2 efflux, and a positive correlation with annual precipitation, attributed to climatic regulation of stream surface area and flushing of soil CO2 (Butman & Raymond 2011).

*4.1.2 Mechanisms responsible for CO2 over-saturation*

CO2 effluxed to the atmosphere can be dissolved biotic CO2 that has been imported into the river system, derived from plant root respiration and decomposition of soil organic matter (Edwards & Harris 1977). However, it can also be the by-product of microbial respiration of organic carbon (e.g., Docherty *et al.* 2006) or in-river respiration by aquatic plants (e.g., Nimmick *et al.* 2011). Measurements show that up to 70% of terrestrial organic matter entering lakes can be respired (McAllister & del Giorgio, 2008).

Abiotic production of CO2 can also occur through photochemical degradation of organic carbon, an important process in the decomposition of detrital organic matter (Mopper & Kieber 2002). For some time it has been thought that natural sunlight levels can cause photochemical degradation of dissolved organic carbon: DOC arriving in the oceans from riverine sources quickly disappears and this was considered largely due to the effect of sunlight on DOC in coastal waters (Hedges *et al.* 1997).

Fig 4.1. The loss of DOC (a), and lignin phenols (d) during decomposition experiments with Broad River water collected in February (open circles and dashed line) and May (open squares and solid line). The % of initial concentrations of variables is plotted as a function of incubation. Time. From Benner & Kaiser (2010).



Organic molecules that absorb light in the ultraviolet region of the solar spectrum are often photoreactive (Benner & Kaiser, 2010). Aromatic molecules, such as lignins (breakdown products of plants which give structural strength), strongly absorb ultraviolet light and break down easily. However, most aliphatic molecules are not very susceptible to photodegradation (Opsahl & Benner 1998, Schmitt-Kopplin *et al.* 1998, Osburn *et al.* 2001).

Experimental analysis subjecting river water rich in C to differing amounts of solar radiation reveals that photodegradation can significantly affect the quantity and quality of carbon losses from a catchment and the component of DOC affected is different to that subject to microbial degradation: photodegradation was primarily responsible for losses of the chromophoric and lignin phenol components of DOM, whereas biodegradation was primarily responsible for the overall remineralisation of DOC and losses of the amino acid component of DOM (Fig. 4.1, Benner & Kaiser 2010). Little work has been carried out in this area with specific respect to peatland carbon sources, but as lignin-derived products will be an important component of the DOM pool, downstream reductions in [DOC] due to uv-photolysis of DOM may be expected.

*4.2 The role of lakes within drainage system carbon cycle*

Lakes are important features of terrestrial drainage systems, playing a central role in the reprocessing of carbon. Their longer residence time compared to river systems allows prolonged respiration of organic matter and additionally the large surface area supports photolysis of organic matter. As such we now recognise that although CO2 in the surface waters can be fixed by algal populations most lakes, especially in carbon-rich catchments with high [DOC] drainage water are over-saturated and emit CO2 (e.g., McCallister & del Georgio 2008). Thus lakes also act as a conduit for terrestrially derived CO2 from the watershed to enter the atmosphere (Cole *et al.* 1994). The processes central to the C-cycle of lakes are summarised in Fig. 4.2 and Table 4.1.



Fig 4.2. The C cycle in lakes (from Tranvik *et al.* 2009). Letters correspond to rows in Table 4.1 below.

*4.3 Burial and sedimentation of carbon in lakes*

Lakes sequester large amounts of sedimentary carbon (Cole *et al.* 2007); it has been estimated that lake sediments contain around 820 Pg of organic carbon (Einsele *et al.* 2001, Cole *et al.* 2007). The global annual amount of sedimentary organic C sequestered in lakes and reservoirs is three times greater than in oceanic sediments (Dean & Gorham 1998). Annual burial rates of organic and inorganic carbon tend to be highest in small, eutrophic lakes and reservoirs (Downing *et al.* 2008). Sedimentary organic C sequestration is high in lakes for several reasons: lakes focus sediment from the watershed (von Wachenfeldt & Tranvik 2008) and retain it; they are generally very productive, generating autochthonous organic matter in the form of algae, and lake sediment oxygen concentrations are often low (Wetzel 2001). Furthermore, as particular fractions of OM are slow to decompose efficient burial can occur (von Wachenfeldt *et al.* 2008). Small and shallow lakes are more common and thus more organic carbon is stored in shallow sediments than deep sediments (Downing *et al.* 2006). There are no estimates of the importance of lakes as carbon storage and processing features in peatland carbon landscapes as distinct from lakes in other types of landscapes.

**Table 4.1** Analysis of the processes that control C concentrations in lakes and associated drivers of change, impact and influence on CO2 efflux. Letters correspond to those used to identify linkages in Fig. 4.2. Adapted from Benner & Kaiser (2010).

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
|  | **Process controlling**  **C cycling in lakes** | **C species**  **affected** | **Example of change**  **driving lake C inputs**  **and processing** | **Potential change**  **in DOC quantity** | **Effect on GHG**  **emission** |
| **A** | Hydrology (e.g., timing and flux of water transporting DOC) | DOC, POC, and DIC | More summer storms driving flow paths through litter layer and wetlands | Increase during summer | More CO2 and CH4 |
| **A** | Water residence time | DOC, POC, and DIC | Prolonged (e.g., North American prairie lakes) or reduced (e.g., Fennoscandia) time for -processing of C within lakes | With increasing residence time, increased loss by photo- and biodegradation | Higher CO2 emissions at high CDOM concentrations (i.e. at short residence times) |
| **B** | Vegetation, CO2 and temperature effects on plant species and litter production | DOC and POC | Increase in the structural, chemical complexity of plant tissue altering bioavailability and degradability | Increase, particularly in the autumn with an increase in leaf litter | Change in CO2 and CH4 flux, more sedimentation |
| **C** | Atmospheric deposition to terrestrial system, more nutrient flux from litter and soils | DOC and POC | More nitrate flux to lakes, driving phosphorus limitation and changes in stoichiometry | More DON | More CO2 and CH4 |
| **D** | Seasonality of lake condition (e.g., ice-cover, flushing, stratification and temporal permanency) | DOC and POC | Increase in ice-free period, driving increase in summer phytoplankton and anoxia-driven methanogenesis in lake sediments | More DOC from algal growth, photolytic loss  of DOC | More CO2 and CH4 |
| **E** | Autochthonous production due to littoral vegetation, phytoplankton, or benthic algal mats | DOC and POC | Change in trophic structure as a result of loss of fish predators | More biolabile non-humic DOC, increase in POC | Less CO2 flux and increase in CH4 from sediment |
| **F** | Microbial DOM degradation | DOC and DIC | Increase in water temperature, driving microbial degradation | Less DOC | More CO2 production in pelagic zone |
| **G** | Photodecay caused by change in irradiance, water quality (pH, iron, and NO3), and stratification | DOC and DIC | Increase in summer UV penetration due to photobleaching of CDOM during prolonged stratification | Less DOC | More CO2 production in  pelagic zone |
| **H** | Sedimentation caused by D salinity, pH, cations | DOC and POC | Drought-caused increase in salinity enhancing DOC flocculation | Less DOC | Less CO2 flux |

If full catchment analysis (section 7) includes a lake in the catchment then unpicking the role of the lake in reprocessing and sedimentation of C is possible through estimates of discharge inflow and outflow and lake volume (to calculate residence time). Clearly there will be a reduction in flow upon discharge into a lake and therefore energy will be lost and POC can be deposited. Estimates of POC export from landscapes into lakes are possible through analysis of dated lake C sediment cores in conjunction with geochemical analysis to identify the source of the C. This approach may be useful in trying to contextualise if modern day POC exports are significantly different from those in the past (e.g., Dickens *et al.* 2011).

**Chapter 5: Guidelines for managing losses of carbon and water through drainage during development activities on peat**

*5.1 Introduction*

To date there have been very few published guidelines on the best way to manage developments on carbon landscapes. Those that do exist are largely concerned with the development of wind farms on peatlands14, with the exception of the Forestry and Water guidelines1 which deal in some part with the management of forestry developments on peat soils. Only the publication associated with the windfarm payback calculator specifically considers carbon losses via drainage from these carbon landscapes14.

Such guidelines have a wide range of general aims. Management of construction or other development activities to limit impacts on drainage (and therefore also aqueous carbon losses) mainly relate to windfarm development15,16,17. However, it is considered that these can form a sound basis in establishing good practice for wider developments. Construction guidelines generally attempt to minimise or mitigate impacts on peatland hydrology.

Mitigation of aquatic carbon losses can also be achieved by using established peatland restoration measures. This area has received considerable and continuing research interest. Much of this concerns the underlying mechanisms and effects of restoration using a variety of tools.

This chapter has two aims: to synthesise the published development guidelines which are intended to reduce environmental impacts in the form of three tables; to complement this with summary discussion of peatland restoration measures.

The guideline synthesis provides details of the recommendation, the recommending body and corresponding documentation, the justification for the recommendation with respect to losses of C via drainage, our comment on this recommendation and whether this recommendation is based on research. Where no scientific justification is provided then we have indicated other studies which might underpin the recommendation. In some instances no other supporting study has been found but we have marked this with an asterisk to indicate our likely interpretation based on current understanding. Finally a qualitative assessment is made of the strength of the evidence underpinning the recommendation e.g., whether direct supporting evidence has been published.

Our synthesis identified three broad areas that existing guidelines focussed on and so we have subdivided the synthesis into these areas:

* Managing excavated or disturbed peat
* Managing the construction process
* Managing water courses and drainage

It should be noted that the recommendations considered here are primarily those relating to drainage and carbon losses. The original documents often consider a wider range of recommendations many of which are outside the scope of the present document.

* 1. *Managing excavated or disturbed peat*

| * 1. **. Managing excavated peat** | | | | |
| --- | --- | --- | --- | --- |
|  | **Recommendation and recommending body** | **Justification and net impacts on losses of C via drainage** | **CLAD appraisal and comment** | **Direct supporting evidence** |
| 1.1 | *Recommendation -* When excavating areas of peat, excavated turfs should be as intact as possible. Often it is easiest to achieve this by removing large turfs.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008) | An intact excavated block will be less prone to drying out.  Intact turfs will help reduce POC loss from surface erosion. | The aim of this approach is to reduce desiccation of extracted peat. Desiccation of peat has been shown to increase C losses as DOC (Fenner & Freeman, 2011; Worrall *et al.* 2005). Extraction as intact blocks will also likely reduce chances of POC production via surface erosion. | No published research. |
| 1.2 | *Recommendation -* Excavated peat soils should be re-used and not discarded or re-worked due to them being saturated. Peat turves require careful storage and wetting to ensure that they remain fit for re-use. When constructing tracks for example, this requires restoration as track construction progresses. However for borrow pits and crane pads this may be more difficult .  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010); Guidance on the assessment of peat volumes, reuse of excavated peat and the minimisation of waste (SEPA /Scottish renewables (2012); Calculating carbon savings from windfarms on Scottish peatlands (Nayak *et al.*. 2008). | No justification provided in provided the study but the aim is to reduce DOC losses to drainage pathways. | This is an attempt to maintain the hydrology of the site. Replacing turves helps preserve the acrotelm within the peat. Wetting them prevents long term C loss by preventing desiccation and subsequent losses of DOC to drainage pathways (Fenner & Freeman, 2011; Worrall *et al.* 2005). | No published research. |
| 1.3 | *Recommendation -* Excavations should be prevented from drying out or desiccation as far as possible. This can be achieved by minimizing disturbance or movement of the excavated peat. Consideration should be given to spraying the peat with water.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008). Guidance on the assessment of peat volumes, reuse of excavated peat and the minimisation of waste (SEPA /Scottish renewables (2012). | None provided in study but the aim is to reduce DOC losses to drainage pathways. | Attempts to reduce dewatering and desiccation will reduce long term losses of CO2. It will also minimise chances of DOC losses via drainage pathways. (Fenner & Freeman, 2011; Worrall *et al.* 2005). | No published research. |
| 1.4 | *Recommendation -* Stockpiling of peat should be in large volumes, taking due regard to potential loading effects to minimise peat slide risk. Piles should be bladed off at the side to minimise the available drying surface area.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008) | No justification provided in the study but on assumption it will reduce drying out and losses of POC, and DOC to drainage pathways. | Attempts to reduce desiccation by reducing surface area for losses via evaporation will reduce long term losses of CO2(\*). It will also minimise chances of POC and DOC losses via drainage pathways (\*).Drained or exposed peat will result in increased aquatic carbon losses (Freeman *et al.* 2001; Wilson *et al.* 2011). | No published research. |
| 1.5 | *Recommendation -* Stockpiles of peat should be minimised.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | No justification provided in the study but on assumption it will reduce drying out and losses of POC, and DOC to drainage pathways. | Large mounds of peat may rapidly dewater causing desiccation and increased rates of c loss via CO2 and DOC. It should be noted that recommendation 1.4 and 1.5 are contradictory. Higher piles are more likely to become dewatered while smaller piles expose a greater area to evaporation. Reducing mound size may also increase likelihood of erosional losses as POC. Overall volumes of stockpiling should be minimised and height and surface areas kept to a minimum – for example against rock faces in borrow pits. | No published research. |
| 1.6 | *Recommendation -* The total exposed ground at any time should be minimised by careful phasing of construction activities.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England 2010). | No justification provided in the study but on assumption that loss of peat mass desiccation, oxidation and wind erosion and export  as DOC and POC to drainage pathways will be minimised | Removed surface vegetation will certainly change the local hydrology and has the potential to result in increased erosional losses(Evans *et al.* 2006))  Losses of acrotelm will further reduce the level of the water table resulting in increased production of DOC and its subsequent loss to drainage pathways (Freeman *et al.* 2001). | No published research. |
| 1.7 | *Recommendation -* Peat stores should not be kept near to drainage channels or on sloping ground.  *Recommending body or bodies and source -* CLAD | This is to reduce the risk of erosional losses of POC to drainage channels. It follows good practice guidelines adopted in agriculture and forestry. | Vegetated buffer zones next to watercourses can filter out suspended solids and reduce input to local streams. Similarly erosion and sediment transport is enhanced on steeper slopes. | No published research. |

* 1. *Managing the construction process*

| **5.3. Management of construction activities** | | | | |
| --- | --- | --- | --- | --- |
|  | **Recommendation and recommending body** | **Justification and net impacts on losses of C via drainage** | **CLAD appraisal and comment** | **Direct supporting evidence** |
| 2.1 | *Recommendation -* Good track design should be employed with appropriate cross drains, minimizing the collection of water on the track  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008). Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010) | To keep track runoff to a minimum and on the assumption that losses of POC to drainage pathways are kept to a minimum | While this measure may reduce particulate losses from the track surface it may alter POC losses from peat areas due to increased flow rates and changed hydrological pathways. | No published research. |
| 2.2 | *Recommendation -* The design of tracks should be such that they do not act as a conduit or channel for water or a dam or barrier to water flow. This requires the consideration of track design in relation to hydrology at the construction stage, when all geotechnical investigations are proceeding, rather than deciding on a final track design at the planning stage. The aim should be to mimic the natural drainage pattern as far as possible.  No ponding of water should be allowed to occur that would lead to additional loadings being placed on the surrounding ground-. This could particularly be the case in areas with peat stability concerns.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008); Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010)/ Good practice during windfarm construction. (SEPA 2010). | To keep track runoff to a minimum and prevent alteration of catchment hydrological pathways on the assumption that it will keep losses of POC to drainage pathways to a minimum.  Justification also given is prevention of soil failure (Peat slippages). This would lead to large losses of POC and possibly DOC. | This measure is directed at maintaining the hydrology of the site as close as possible to that which existed pre-development. The instruction is to ensure the track does not act as a channel for water should decrease the risk of erosional losses. A suggested effect of this measure is reduced risk of creating new drainage channels with associated losses of POC (Evans *et al.* 2006).  Peat slippages due to saturation remove large volumes of surface peat (Dykes & Jennings 2011, Warburton 2004) and have the potential to increase C losses as disturbed areas of the acrotelm dewater and decompose. Also, exposed areas of peat have the potential to experience increased erosional losses of POC (Evans *et al.* 2006) | No published research.  Warburton *et al.*. (2004) |
| 2.3 | *Recommendation -* Floating roads should be used where peat is deeper than 1 m to avoid cutting into peat and disturbing it leading to drying out. Cut roads should only be used in areas where the peat is less than 1 m deep.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008) | No justification provided in the study but on assumption that disturbance to hydrology will be minimised and DOC and POC losses kept to a minimum. | Cut areas are exposed and have the potential to increases delivery of POC to drainage channels (Evans *et al.* 2006).  Cut areas will also require associated drainage which is known to be associated with increased losses of DOC and POC (Fenner & Freeman 2011; Worrall *et al.* 2005).  Disruption of the ‘normal’ local peat hydrology has the potential to reduce the water table and increase losses of DOC and CO2. | No published research. |
| 2.4 | *Recommendation -* Install offlet culverts and ditch blockers to prevent the build- up of flow along permanent track edges . Where not possible settlement lagoons may be needed. It is important that the track surface remains free from standing water, and that any collected water filters across a vegetated buffer zone, or through a settlement lagoon, before reaching any watercourses, lochs, groundwater or sensitive wetlands.  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA 2010) | Track edge drainage controls run-off from the road surface and from upslope to downslope areas. The purpose of offlet culverts are to reduce the potential for a larger volume flow down the ditch.  . | The purpose of these measures is to control flow pathways and water velocities. Flow reduction lessens the potential for track edge erosion during periods of high rainfall and POC losses. Allowing the flow to spread out reduces velocities and erosional risk. | No published research. |
| 2.5 | *Recommendation -* Maintain tracks to avoid rutting.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010) | No justification provided but on the assumption it will reduce POC losses. | Rutting may produce new flow pathways and volumes leading to potential erosional losses of POC (\*). | No published research |
| 2.6 | *Recommendation -* Install surface cross-drains on long track segment with a substantial gradient to intercept runoff running down the road. These cross-drains can be constructed with channels of various materials but should be strong enough to withstand the expected traffic loadings.  ­*Recommending body or bodies and source -* Good practice during windfarm construction (SEPA 2010); Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010) | Cross drains intercept surface flows, and divert then into the side ditches, preventing the build-up of flow and reducing POC losses | While this measure is intended to reduce particulate losses from the track surface it may also reduce POC losses from peat areas to increased flows travelling along newly formed flow pathways (Evans *et al.* 2006). | No published research |
| 2.7 | *Recommendation -* Prevention of ponded water upgradient of tracks by placement of culverts or pipes. These need to be of suitable dimensions to accommodate floodwaters and prevent sediment build up behind them. Culvert invert levels should be slightly lower than the inlet and outlet levels to allow a natural bed to form within them.  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA 2010) | The justification is the maintenance of the natural hydrology. Overhanging outlets should be avoided as this will cause erosion and increase POC losses | These measures may help to limit losses of POC caused by erosion due to changed flow pathways (Evans *et al.* 2006) | No published research |
| 2.8 | *Recommendation -* Track drainage designed to intercept large volumes of water should be porous to minimise direct discharge to watercourses.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | To minimise direct discharge to watercourses and reduce losses of DOC to drainage pathways | This measure will help recharge groundwater keeping the water table high and reducing losses of DOC caused by faster drainage rates.(Freeman *et al.* 2001,\*) | No published research |

* 1. *Managing water courses and drainage*

| **5.4. Management and Monitoring of Drainage** | | | | |
| --- | --- | --- | --- | --- |
|  | **Recommendation and recommending body** | **Justification and net impacts on losses of C via drainage** | **CLAD appraisal** | **Supporting evidence** |
| 3.1 | *Recommendation -* Developers should take ancillary opportunities to improve peatland habitat, by including practices such as drain blocking and re wetting of areas in their management plan.  *Recommending body or bodies and source -* Calculating carbon savings from windfarms on Scottish peatlands – a new approach (Nayak *et al.*. 2008) | These practices can be included as mitigation activities in that in theory they should reduce DOC and POC export to surface water. | Drain blocking and re-wetting has the potential to create the conditions for sphagnum growth and therefore peat development and C sequestration. Blocking also raises the water table leading to reduced losses of DOC (Wilson *et al.*. 2010). | See reviews in section 2 below |
| **3.2** | *Recommendation -* Engineering activities such as culverts, bridges, watercourse diversions, bank modifications and dams are avoided wherever possible. Where watercourse crossings are required, use bridging solutions or bottomless arched culverts which do not affect the bed and banks of the watercourse are recommended as they do not impact so much on fish passage and movement of animals such as otters.  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010). | The justification is to avoid hydromorphic and ecological modification of the water environment or enhance POC export. | Alterations to the morphology of the site and disturbance of riparian margins are likely to result in increased production of DOC and POC. | No direct research |
| 3.3 | *Recommendation -* Produce sediment and erosion control plan. Ensure that all contractors on the site understand and comply with the plan. Revise the sediment control plan if new sources of sediment are identified during construction.  Monitor water quality in key watercourses to ensure sediment load does not exceed suitable limits.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010) | This should minimise sediment losses at source or have in place sediment retention devices.  To ensure carbon losses are minimised and to identify when mitigation methods for suspended solids loss are not working. | Identifying potential sources areas, avoidance and management at the planning stage can reduce sediment losses and avoid siltation problems at later stages of the development phase.  Automatic water sampling and use of turbidity meters are required with a telemetry alarm system to notify when turbidity levels are high. | Not applicable |
| 3.4 | *Recommendation -* Both temporary and long term foul drainage provisions and maintenance considered and authorisation sought if applicable.  *Recommending body or bodies and source -* Good practice during windfarm construction(SEPA, 2010) | That this can compound water quality issues | No comment required. | Not applicable |
| 3.5 | *Recommendation -* Avoid any barriers (e.g., bunds; bentonite walls) to surface or groundwater flow.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England , 2010) | No justification provided but on the assumption that losses of DOC and POC via will be minimised. | Hydrology is a significant control on site C fluxes. Disturbing hydrology has significant risks for increased on site carbon flux. | No direct research |
| 3.6 | *Recommendation -* Monitor groundwater levels and rainfall during construction. Avoid activities which may result in instability during very wet times.  Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010) | No justification is provided | Periods of high runoff can lead to a more sensitive catchment in terms of DOC and POC export. | Not applicable |
| 3.7 | *Recommendation -* On afforested peatlands prior to clearance identify and assess all watercourses, and plan restoration work to reduce the risk of erosion and sediment delivery.  *Recommending body or bodies and source -* Forestry and Water (Forest commission, 2011) | The justification is reduction of the risk of erosion and sediment delivery in the catchment and thus reduced losses of POC | This measure is pertinent to felling of plantations on peatlands and should help reduced erosional losses of POC | No published research relevant to DOC export but studies on suspended sediment by the Institute of Hydrology and others occurred in 1970s and 1980s. |
| 3.8 | *Recommendation -* Pre-earthworks drainage should be undertakento minimise the effects of pooled water on the stripped/exposed soils once earthworks commence. This should be installed on the “high-side of areas that are affected by track construction operations. Drainage ditch profile can vary from a ‘V’ to a ‘U’ shape. “V” ditches tend to maintain more vegetation as their footprint is less than a “U” ditch but are generally successful in harder ground conditions that would not be susceptible to erosion. On completion of the works can sometimes be backfilled. If at all possible, any stripped turves should be placed back on the sides of the ditches to assist regeneration, and also to reduce potential erosion in softer soils.  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010). | To intercept stormwater surface run-off. This prevents it “ mixing with “construction” drainage.  Dedicated piped culverts significantly reduce volumes of potentially sediment-laden water that would need further treatment. Overall this should reduce POC export. | This measure should reduce POC produced by erosion and minimise its loss to the natural drainage network. | Evans *et al.*. (2005, 2009) |
| 3.9 | *Recommendation -* Silt Traps should be installed on the inlet or outlet side of culverts, but require to be robust enough to allow for frequent clearing out of accumulated sediment .  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010). | Silt traps can be a simple and effective method of controlling sediment laden run-off and reduce losses of POC to downstream water bodies. | Silt can contain large amounts of carbon originating from the peat and preventing its loss may be considered a carbon saving (Kay *et al.* 2009). Thought also needs to be given to the fate of the collected sediment to ensure that cleaning of traps and storage of the deposited sediment does not create their own sediment-related issues. | No published research. |
| 3.10 | *Recommendation -* The installation of silt fencing on exposed slope. These are often made of semi-permeable geotextile fabric, vertically held on timber posts, and used as a means of filtering sediments.  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010). | Silt fencing may trap suspended sediments and retain it on site. As such it reduce losses of POC to the stream network | Silt can contain large volumes of carbon originating from the peat and preventing its loss may be considered a carbon saving. The degree to which they may be effective is not fully unquantified but trials are underway in Scotland on agricultural catchments (e.g., Lunan water catchment being conducted by the James Hutton institute). | No published research |
| 3.11 | *Recommendation -* The installation of Straw Bales on watercourses  Installation positions require careful consideration as they should allow for overtopping during high flow. Bales that become silt-laden and need replacing to be effective require to be discarded and are subject to waste legislation  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010). | Straw bales can be used to filter out sediments from normal flows in drainage ditches and hence reduce losses of POC. | This measure may potentially remove POC and prevent C losses in this form. Heather bales have also been used as a means of drain blocking and retention of sediment. The degree to which they may be effective is not quantified as yet. (Armstrong *et al.* 2010). | Armstrong *et al.*. (2009) found that straw bales were not always effective at sediment trapping. They also noted concerns regarding foreign seeds and nutrient export. |
| 3.12 | *Recommendation -* Use of settlement lagoons, ponds and silt traps and treat runoff in line with best practice. Lagoons are effective where a large run-off volume is expected and small scale dispersal to existing vegetation would not be successful. Large pools should be compartmentalised to allow progressive filtration and settlement to occur, from the inlet to the discharge end.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010); Good practice during windfarm construction (SEPA, 2010). | No justification was provided in study but obviously a sensible recommendation.  Reduced losses of POC via drainage pathways. | Settlement pools are considered to be an important potential intervention to prevent losses of POC. | No published research. |
| 3.13 | Flocculant dosing is last resort to induce settlement of sedimenst where mitigation methods have failed. Liquid flocculants can be dosed into settlement lagoons, and solid flocculant blocks can be set in flowing water to slowly dissolve, thereby giving a prolonged ‘dose’.  *Recommending body or bodies and source -* Good practice during windfarm construction. (SEPA, 2010). | Flocculants works by attracting together fine particles in to larger heavier aggregates that settle out quicker. This will reduce POC losses. | Forced settlement of POC (Cathalifaud *et al.* 1998) has the potential to reduce losses of carbon via drainage pathways (\*) | No published research. |
| 3.14 | *Recommendation -* Where there is a significant build-up of water or in turbines bases during construction pumping may be necessary to avoid further pooling or to allow works to progress. The pumped water discharge should be directed to a settlement lagoon or in to natural vegetation well away from any streams, lochs, or wetlands (avoid adding water to areas of peat > 0.5m depth). Equipment such as “Siltbuster” type tanks can assist with suspended solid reduction  *Recommending body or bodies and source -* Good practice during windfarm construction (SEPA, 2010); Natural England Assessing impact of wind farms on Blanket Peat in England (2010) | None provided in study but is assumed that this guideline is given to reduce risks of peat slippage/slides through excessiveloading and reduce losses of POC and DOC associated with peat slippage. | No comment | No published research. |
| 3.15 | *Recommendation -* Align forest drains to run at a maximum gradient of 2 degrees (3.5%) and lead them towards the heads of valleys.  *Recommending body or bodies and source -* Forestry Commission Forestry and Water (2011). | There was no justification provided in the study but presumably on the basis that it would reduce losses of POC. | Drains of high gradient are prone to erosion (Holden *et al.*. 2007). A value of 3 degrees has been widely quoted as a critical angle on peatlands. | Research on this was undertaken by the Institute of Hydrology and others in the 1970s and 1980s. |
| 3.16 | *Recommendation -* Drainage ditches should have low gradients, depths should be minimised as should the length of drains both individually and in total. Check dams / erosion protection in ditches with slope greater than 5% should be installed.  *Recommending body or bodies and source -* Natural England Assessing impact of wind farms on Blanket Peat in England (2010). | There was no justification provided in the study but limited water table drawdown and flow velocities should minimise the mobilisation and transport of DOC and POC. | Drainage lowers the local water table allowing more oxygen into the acrotelm and results in increased decomposition rates (therefore greater C flux) (Freeman *et al.* 2001; Wilson *et al.* 2011) and greater losses of DOC and POC. Large discharge increase the erosive power of the water and can result in greater losses of C as POC.(Armstrong *et al.* 2010; Holden *et al.* 2007) | Drainage has been shown to be associated with increased DOC losses . (Tipping *et al.*. 1999, Wallage *et al.*. 2006) and POC losses Holden *et al.* Armstrong *et al.* (2009)  Recommended differing management for ditches with slopes of >3o. |
| 3.17 | *Recommendation -* Camber tracks to avoid ponding and maximise track runoff.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | The justification is to avoid build-up of water and erosion potential. | This may be particularly important where the track is on a slope and otherwise water will run down the track and lead to erosion. | No published research |
| 3.18 | *Recommendation -* Construct tracks from coarse particle size aggregate  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | The justification is coarser material less likely to be eroded and transported and more permeable minimising runoff. | No comment. | No published research |
| 3.19 | *Recommendation -* Creation of frequent surface water discharge points via level spreaders.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | The justification is that this will avoid the build-up of water and possible erosion risk. | No comment. | No published research |
| 3.20 | *Recommendation -* Backfill cable trenches and minimise the amount of time they are open. Use clay bunds in backfilled trenches to prevent any flow of water along their length.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | There was no justification provided in the study but presumably on the basis that it would reduce losses of POC and DOC. | Cable trenches will act as temporary drainage channels and may reduce the water table locally resulting in increased DOC production and export (Freeman *et al.* 2001, Wilson *et al.* 2011). Erosion and high levels of POC transport may be produced along their length during high flows | No published research |
| 3.21 | *Recommendation -* Check crossings for blockages especially during and after heavy rainfall.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | There was no justification provided in the study but presumably on the basis that it would reduce losses of POC. | Using the precautionary principle it is suggested that changes to ‘natural’ hydrology (in this case flow pathways) have the potential to disrupt site C flux. | No published research |
| 3.22 | *Recommendation -* Channel runoff from hard standing areas.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | There was no justification provided in the study but presumably on the basis that it would reduce losses of POC. | Large volume fast flows on roads have the potential to increase erosion on peat at the point which they leave the road creating increased risk of erosional POC loss. Managing this flow may reduce POC losses. | No published research |
| 3.23 | *Recommendation -* Construct water crossings to reduce flow at either end and use edge constraints to reduce splatter from vehicle wheels.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | Potential to reduce bankside erosion.  Reduced losses of POC via drainage pathways. | This measure has potential to reduce losses of POC created by abrasion of vehicle wheels on the edges of the drainage channel. | No published research |
| 3.24 | *Recommendation -* Avoid uncontrolled discharge of water over peat surfaces.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | There was no justification provided in the study but presumably on the basis that it would reduce losses of DOC and POC. | This measure has the potential to reduce losses of erosional POC associated with high velocity flows (\*) and risk of peat slippage for large discharges on slope areas (Dykes & Jennings, 2011, Warburton *et al*. 2004) | No published research |
| 3.25 | *Recommendation -* Avoid soakaways, SUDs or drainage into areas of deep and/ or potentially unstable peat. Use surface level spreader.  *Recommending body or bodies and source -* Assessing impact of wind farms on Blanket Peat in England (Natural England, 2010). | There was no justification provided in the study but presumably to reduce the risk of peat slides. | This measure is designed to reduce the risk of peat slippage which is likely associated increased local losses of POC and DOC via drainage pathways (Dykes & Jennings 2011; Warburton *et al.* 2004). | No published research |

* 1. *Peat restoration as a mitigation measure*
     1. *The importance of restoring peat to carbon storage and losses*

Peatland restoration programmes are used to address peat degradation globally. Given the focus of this document, we confine our discussion to the hydrological component of peatland restoration techniques, but it should be noted that hydrological modifications may be part of a wider suite of approaches for successful restoration.

The approach to peatland restoration is somewhat dependent on hydrology and size of the drainage network. However, common aims of restoration projects are (i) to raise the water table through ditch, gully or river blocking, (ii) to revegetate the site to minimise soil erosion and/or (iii) to manage the vegetation for example by controlling non-native species. A survey of 56 UK peatland management and restoration projects (Walker *et al.* 2008) found that restoration or maintenance of biodiversity was the primary objective for all projects. Hydrological function and carbon storage were stated as the second and third most important justifications. With more focus on the importance of maintaining terrestrial C stores, these priorities may change.

Raising the water table is a key component of peatland restoration. Without this, an artificially lowered water table under past management allows oxidation of the peat leading to increases in CO2 efflux and/or increases in aquatic export of carbon. The following draws from Holden, (2009) and summarises the key components of ditch-blocking with respect to the consequences for drainage and aqueous carbon losses:

1. Surface flow velocities across drain-blocked peatlands are slower than in their drained counterparts, thought to be due to the more frequent and greater presence of *Sphagnum* moss cover around blocked drains. Flow velocity is also slower in vegetated drains (even ones without dams and pools) compared to bare peat drains by at least 10-fold (Holden *et al.* 2008a). This has the potential to reduce export of POC which is produced by the erosion caused by water movement.
2. For the same catchment area, the volume of water transported by blocked drains is significantly reduced relative to unblocked drains (Holden, 2005a). Recovery of the mean water table may be relatively rapid. However, water table dynamics remain disturbed for some years (displaying a greater range of depths) after blocking relative to sites with intact peats and are controlled by different processes - evapotranspiration in intact peat and water movement through the drained peat.
3. Drain-blocking in blanket peat usually leads to a decrease in water colour and DOC concentrations in peat (Wallage *et al.* 2006) and drainage channels. For example, an extensive survey of 32 sites (Armstrong *et al.* 2010), comparing blocked with unblocked drains where both still had flowing water, found that [DOC] was significantly lower in blocked drains (mean difference of 28% less) - and so we can infer in the drainage run-off. However there are times when no significant difference in [DOC] can be found between drained and blocked systems (e.g., Gibson *et al.* 2009, Jonczyk *et al.* 2009, Worrall *et al.* 2007). For example intensive monitoring of a peat drain system that had been blocked for 7 years showed that whilst [DOC] in unblocked drains was greater, the annual export per unit area of catchment was comparable to the blocked drain (Armstrong *et al.*, 2010). This has implications for critical losses (section 6.4).
4. Ditch-blocking changes the composition of peatland DOC (e.g., Gibson *et al.* 2009, Jonczyk *et al.* 2009, Wallage *et al.* 2006).
5. Vegetation change (driven by management practice, including drain-blocking) may have a strong impact on dissolved organic carbon production (e.g., Holden 2009, Wilson 2011).
6. Drain-blocking has been shown to reduce fine sediment and thus POC (e.g., Wilson *et al.* 2011).

Without restoration, if net losses are greater than sequestration rates the peat body will lose mass. If drying of the surface leads to loss of vegetation, channelling of surface run-off is more likely. This can lead to the formation of peat gullies. Gullies can be also formed through natural (e.g., climate-induced) changes in peat water and vegetation balance, or through disturbance by catastrophic events such as fire, or chronic impacts such as overgrazing or excessive trampling. However, once formed, gullies support continued erosion during runoff events, triggered by high intensity rainfall or prolonged rainfall on saturated ground. Thus gullies are responsible for export of large amounts of peat, contributing to large loads of POC in watercourses (Evans *et al.* 2006).

Techniques such as the construction of dams from natural materials (peat, stone and plastic) in the heads of gullies have the potential to reduce sediment loss, raise water tables and help re-establish vegetation (Evans *et al.* 2005). In this study, dams were constructed near the heads of gullies, below a maximum slope angle and within an optimum height range of 25-45 cm. Spacing of the dams was sufficient to allow pooling during peak flow enabling overspill of each dam to fall onto a water surface and so reducing the potential for erosion (Evans *et al.* 2005).

The most common (and generally cheapest) approach to the restoration of peatland hydrology is to block ditches and channels within the artificial drainage system (e.g., Worral *et al.* 2007, Armstrong *et al.* 2009). Ditch profiling may be required where the lower part of the ditch has become eroded (Armstrong *et al.* 2009). In the UK artificial drainage tends to be < 0.5 m wide and so these ‘ditches’ are blocked, often with blocks of intact peat, at regular intervals along their length. This allows water tables to rise and the surface to be more water-logged slowing down decomposition of organic matter and creating the wet conditions for *Sphagnum* regeneration.

* + 1. *Informative approaches to restoring peat bodies*

There have been a number of recent reviews and overview documents on the benefits or otherwise of drain-blocking. For example, a manual exists for restoration of Canadian peatlands, including guidance on ditch-blocking techniques (Quinty & Rocheforte 2003) ; there is a global guide to the restoration of peatland habitats (Joosten & Clarke, 2008) and the effects of ditch-blocking as part of a comprehensive investigation into the effects of peatland restoration (Lunt *et al.* 2010); a review of the impact of ditch blocking on hydrology and carbon losses was commissioned by the Environment Agency (English regulatory body; Holden 2009); DEFRA (another UK environmental agency) commissioned a review on the potential for ditch blocking to increase methane emissions (Baird *et al.* 2009) and subsequently research to investigate and provide mechanistic explanations for methane emissions.

Perhaps one of the most accessible approaches to drain-blocking has arisen from an extensive survey of UK drained peatlands across a latitudinal gradient in order to assess the most effective approaches in ditch blocking (Armstrong *et al.* 2009). The resultant decision tree (Fig. 5.1) represents a strong evidence-based approach to inform the practical aspects of ditch blocking currently. Advice is given on the most appropriate materials to use and the circumstances and methods in which they should be deployed. This tool asks the practitioners to examine factors including:

* ease of access for machinery
* personnel available
* distance to walk in to the site
* angle of slope on which the drain is located
* wetness and dryness of the peat
* exposure or otherwise of mineral substrate
* the cross-sectional area of the drain
  1. *Towards a better understanding of the effectiveness of restoration on aquatic C losses.*

As more effort is invested in peatland restoration, either through mitigation associated with development or more directly to restore a degraded peatland system (possibly as part of a carbon credit scheme) there will be opportunities to further our understanding of the drainage response. Critically this could be hindered if data are not shared with the wider scientific community to enable researchers to learn. Such a scenario occurs frequently where data are collected by a private company on behalf of a developer and where use of these data to promote further understanding of generic issues is considered commercially sensitive.

We would therefore suggest that any guideline associated with restoration of peatland should promote the idea that data collected should be made public to allow others to learn about best practice and maybe adjust their own practice accordingly. Additionally there seems to be less research on the effectiveness of blocking on the lower catchment carbon export budgets, and maybe measurement here is more robust that within the drains where very local processes could control temporal variations. Thus adding downstream sampling to a drain-blocking research programmes would be valuable and further our understanding of the effectiveness of this activity.

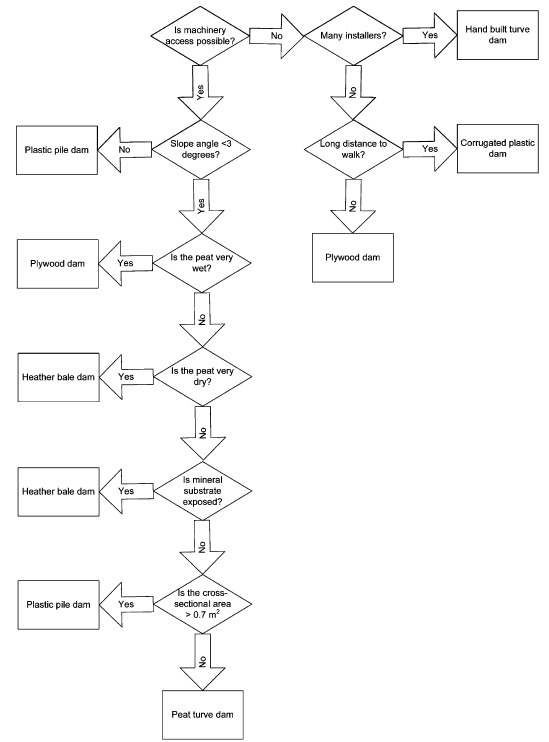


Fig 5.1. Decision tree on drain blocking to aid the peat restoration practitioner (from Armstrong *et al.* 2009)

**Chapter 6: Future challenges**

As important terrestrial stores of carbon, carbon landscapes occupy an important place in the Earth’s carbon cycle. Peatlands are particularly important as slow to capture and sequester carbon due to their relatively low productivity rates. However, those that have remained undisturbed over long periods of time (millennia) have accumulated very large carbon stores. A primary concern for decision makers is that stores such as these should not be degraded and their carbon returned to the atmosphere. In this context, there are many challenges which stakeholders face as they seek to ‘manage’ carbon landscapes. Here we discuss two important areas relevant to drainage: the first is environmental responses to a changing climate; the second is identifying from catchment drainage whether there is a critical loss of carbon from the landscape and how this may be considered within a conceptual (legal) framework to manage these losses.

*6.1 Environmental responses to a changing climate*

There are many publications on projected climate change on carbon landscapes (e.g., soils: Smith *et al.*. 2008; Canadian peatlands: Moore *et al.*, 1998; tropical peatlands: Li *et al.*. 2007; UK peatlands, Clark *et al.*. 2010; generic discussion: IUCN-Worral *et al.* 2010 ). While the potential qualitative risks to peat in a changing climate are known (e.g., potential shifts from net carbon sink to source as decomposition rates increase due warmer drier conditions), the extent to which climate change may affect carbon losses via drainage pathways is unclear. Similarly, more subtle questions of inter-regional differences in impact are also unknown, and here scale becomes important. For example, climate change projections for the next 100 years in the mainland UK suggest an increase in mean temperatures, potential shifts in the distribution of precipitation, warmer and wetter winters and warmer and driers summer - but there are latitudinal gradients (Hulme *et al.* 2002, Jenkins *et al.* 2009) and the control of these can be shown in forward projections. For example modelling the potential impacts of a projected future on UK peat distribution revealed that the bioclimatic envelope for new peat initiation was pushed towards NW Scotland from the current wider UK range, as it was only here that suitable temperature and hydrological conditions were predicted to occur (Clark *et al.*. 2010).

Projected changes can be identified by using process-based models (e.g., Belyea & Malmer, 2004; Wu 2012), or generalised linear models with special modules to describe terrestrial systems, to run the models under IPCC special emission scenarios (IPCC, 2000). The output is compared to the current landscape or biogeochemical response e.g., how peat accumulation rates may change. At local scales such an approach reveals uncertainty in the projected response. For example, the outputs of 12 different models (of two different types: dynamic process-based and bioclimatic envelope) used to project bioclimatic controls on peatland distribution for the UK indicate that, although existing peatlands could persist for decades under a changing climate, there is likely to be a long-term decline in the distribution of actively growing peat, especially if greenhouse gas emissions remain high. However, the results varied between models, as well as between sites (Clark *et al.* 2010). The output depended on whether the model was most sensitive to temperature, in which case there was retreat in peat initiation towards high altitude northern UK, or to water deficit, which was manifest as retreat towards the projected wetter north and west (Clark *et al.* 2010).

Although there is uncertainty and such uncertainty can be critical (for example, where there is not convergence on whether net C sequestration will occur), this largely affects the detail of model outputs and in general projected outputs of models are similar. For example, modelling of increasing summer temperatures in the UK projected declining spatial extent of peatland, albeit to different extents, controlled by changes in bioclimatic space (Gallego-Sala *et al.* 2010). This indicates confidence that despite such variation we still need to consider the impact of the important controls on plant productivity-degradation and thus soil carbon sequestion and the carbon cycle: temperature (e.g., Hossell *et al.* 2000) and hydrological regime (e.g., Finland, Parviainen & Luoto 2007).

Given the wealth of climate-oriented studies that exist and that projections are continuously being refined (e.g., Frolking *et al.* 2011), it is not so critical we outline global detail for projected climate change; instead we focus on temperature and hydrological regime controls relevant to drainage. These are:

1. temperature change is projected to be higher in the N latitudes where many important terrestrial C stores exist;
2. this may cause change to permafrost and so there could be increases in C export in drainage systems draining permafrost regions (Frey & Smith 2005);
3. temperature changes may induce a longer growing season and milder winter and this may increase the time period of DOC production;
4. in turn this may change the autumn DOC flush regime in peatlands in the temperate zone but this timing is also controlled by the hydrological cycle.

*6.2 The changing hydrological cycle*

The importance of hydrology in controlling carbon flux has been demonstrated in Chapters 3 and 4. There are two important aspects of a changing hydrological cycle that will impact on the concentration of C in rivers reflecting the load exported from a landscape, reduced rainfall (drought) and more intense rainfall (flooding):

*6.2.1 Reduced rainfall (drought)*

In areas such as the UK, the frequency and length of drought events is projected to increase (IPPC reference). Many studies have linked drought events and their aftermath to extreme (both large and small) DOC concentrations (Freeman *et al.* 2004, Worral *et al.* 2005, Monteith 2007). Recent experimental work has shown that drought has the specific effect of increasing C losses from peatlands primarily driven by the changes in hydrology (Fenner & Freeman 2010). Drought stimulates bacterial growth and phenol oxidase activity, resulting in a reduction in the concentration of phenolic compounds in peat. This stimulates more microbial growth, resulting in the breakdown of organic matter and production of CO2. Rewetting of the peat following a drought accelerates carbon losses to the atmosphere (Fig. 6.1) and receiving waters (as DOC) owing to drought-induced increases in nutrients and labile carbon. The increase in C losses following a drought can last years after the initial drought event that triggered it. This increase in losses is exacerbated as the peat is rewetted stimulating further bacterial degradation and is sustained even in post drought years.



Fig 6.1. Effect of the 2006 severe natural drought (water table 30 cm below surface) on oligotrophic peatland net CO2 flux. CO2 losses increased during the drought but accelerated during the rewetting phase. Light and dark shading denote replicate 2 different wetlands sampled. Five sampling stations averaged over distinct four-month periods (before, during and 1 year after the event, at 10 cm depth) is shown. Error bars denote standard error of the mean. From Fenner & Freeman 2011.

Peatland managers may well wish to consider the extent to which future droughts impact upon the carbon balance of the peatlands for which they are responsible. This may be particularly important for areas which are identified for the sale of carbon credits (e.g., Dunn & Freeman, 2011). Additionally, prolonged increased DOC export is likely accompanied by increased water colouration so this response may present challenges for water abstraction catchment management. The development of drought avoidance or mitigation measures may also be considered a priority in safeguarding these areas. Given the slow rates of C sequestration in soils is not easily measured and the logistical challenges in measuring gas efflux from large area, catchment monitoring programmes may be useful to assess if intervention management is contributing positively to retaining C in the landscape.

*6.2.2 More intense rainfall (flooding).*

It has been projected that under a changing climate, the magnitude and frequency of flood events will increase in the UK and elsewhere (The UKCP09). As maximum DOC and POC export from catchments takes places at high flows (Clark *et al.*. 1997) increases peak discharges and frequency of events may alter the pattern of DOC export and thus total loads. Ultimately however DOC and loadings are controlled by carbon production and in the event of carbon production being limited at high flows, a reduction in [DOC] may be apparent, although total export would remain high. In high magnitude flood events the proportion of saturation excess flow pathway water, low in DOC, is likely to be higher. This emphasises that when viewing DOC fluxes from catchments it is important to understand hydrological pathways and the relationship between, concentrations, discharge and load.

Increased precipitation intensity, runoff and flood peaks under climate change are likely to increase POC loss from catchments due to overland flow being more widespread and frequent, and shear stresses within in-channel flow higher. The effect of the increase in erosive potential is likely to be particularly prevalent in disturbed peatlands. Unvegetated ground, stockpiles of peat and bare banks in natural and artificial drainage channels will be more susceptible to erosion from overland and channelized flow. In particularly scour of river channels and eroded peat blocks (Evans & Warburton, 2001) are likely to be a more significant occurrence.

Whether increased run-off can remove more C from the terrestrial stores such that net sequestration is affected, or the balance of C lost as gaseous emissions is altered is largely unknown. The latter is important as a feedback to a changing climate. For example, DOC removed by high flow and degassed from the surface water to the atmosphere as CO2 will have a lower global warming potential (GWP) than if further reduced to CH4 and emitted from the carbon landscape in this form. Thus a challenge of assessing net changes to C export from a terrestrial C store is to design and implement a sampling programme that accommodates high flows. This is discussed further in Chapter 7.

*6.3 The challenge of determining ‘critical losses’ and ‘critical loads’ of landscape C export.*

Terrestrial systems that sequester C have always lost a component of this carbon to the atmosphere and drainage systems. The longevity of terrestrial C storage under projected changes in temperature and hydrological regime needs to be considered in the context of whether any net landscape loss is critical. The key questions are:

* How viable would marginal peatlands be under a changing climate?
* At what point is carbon lost faster than it can be sequestered?
* In the absence of conditions needed to support a stable peat habitat should decision makers continue to manage these areas as peatland or invest in other carbon landscapes?

At present whilst there have been individual academic studies on net carbon balance of carbon landscapes (e.g., Billett *et al.* 2004, Nilsson *et al.* 2008) and strong cases made for the need to conserve peatlands (e.g., IUCN, 2012) no framework for assessing critical losses from carbon landscapes has yet been outlined. Producing such a consensus framework is a significant challenge and a key component of this will be accurate assessment of exported aqueous carbon loads (discussed further in chapter 7).

It may be that an existing framework for critical loads will provide some direction. For example, a result of two decades of scientific research, including major research programs in Europe (the Surface Water Acidification Programme ended 1990) and United States (the National Acid Precipitation Assessment Program) we have a definition of critical loads for acid deposition (Nilsson & Grennfelt, 1988) which could possibly applied in a modified form to critical losses of carbon:

"A critical load for acid deposition is the highest deposition of acidifying compounds that will not cause chemical changes leading to long term harmful effects on ecosystem structure and function."

The critical load of acid deposition is an interesting example to expand from as there are parallels with aquatic C losses: acid rain is widespread, there are long recovery times and there can be deleterious effects on aquatic communities. Furthermore, DOC exports have been shown to increase with reduced acid deposition (e.g., Monteith *et al.* 2007; Evans *et al.* 2012). Additionally several studies have examined the degree to which buffering by DOC acids has impeded the recovery in pH (Driscoll *et al.*. 2003, Evans *et al.*. 2008, Erlandsson *et al.*. 2010), but none have addressed the question of how DOC initially influenced acidification, because the historical reference conditions against which anthropogenic acidification may be compared have not been determined (Erlandsson *et al.* 2011). Hindcasting the DOC concentrations of surface waters prior to industrialisation is possible if pH is known (Erlandsson *et al.* 2011).

While the role of DOC in the historical acidification of surface waters is yet to be elucidated, it is suggested that the critical load concept may be modified with respect to DOC and other forms of carbon export. Here it can be further sub-divided into a definition relevant to carbon sequestration (a critical loss if this is reduced) and a definition for the water quality of the drainage system (a critical load if losses are increased):

* 1. Carbon sequestration: "An annual critical loss is the minimum loss of C from a terrestrial landscape that is feasible whilst still maintaining net C sequestration (ie the landscape remains a net C sink)".
  2. Water quality: "A critical load for DOC influx to catchment drainage is the highest input of DOC to the drasinage system that will not cause physiochemical changes leading to long term harmful effects on the aquatic ecosystem structure and function."

One legal framework which might be adapted to allow for the management of aquatic carbon losses is the European Union Water Framework Directive (WFD), within which many other issues of water quality are subsumed. The WFD stipulates that ecological disturbances should be defined by comparison with a reference condition that represents the undisturbed state. This legislation raises the potential opportunity to adopt the concept of critical losses as a framework within which to view carbon losses from drainage pathways. Full carbon budgets of carbon landscapes are required to calculate critical C losses; only catchment drainage needs monitored to calculate a critical load. Regardless, as we move towards increasing need to understand the impact of changes in the C cycle, understanding the dynamics of the catchment drainage appears critical.

**Chapter 7: Monitoring and assessment of aqueous carbon losses**

* 1. *Introduction*

The motivations for investigators determining aquatic carbon losses vary from the applied e.g., to understand better which sub-catchments contribute most colour to abstracted water subject to expensive water treatment, to the more research-led e.g., what are the sources, pathways, drivers and mechanisms of terrestrial carbon loss (critical losses, section 6.4). Whatever the motivation, this chapter introduces a number of issues of interest to investigators considering the practicalities of quantifying organic aquatic carbon losses from carbon rich landscapes. Both direct and indirect approaches to determining dissolved and particulate carbon concentrations are presented. Firstly we present methods for manual collection of water samples for carbon concentration determination, then for automated water quality monitoring, laboratory analysis of collected samples and specific equipment needs. Data analysis to construct a carbon budget for a catchment is then considered, with particular consideration of which timescale (flood, daily, seasonal, annual) is best to quantify carbon export. Finally we discuss the practical and logistical considerations that need to be addressed when designing a sampling strategy to determine baseline carbon losses and produce flux estimates.

* 1. *Determination of C concentrations*

DOC and particulate carbon concentrations can be determination using two approaches: field collection of water samples and subsequent analysis in the laboratory, or in-situ monitoring using proxy indicators of aqueous carbon. Here we focus first on manual sampling and subsequent laboratory analysis and later will discuss proxy indicators.

* + 1. *Sample collection and storage pre-analysis*

There are various manuals that offer guidance on this approach (e.g., Wershaw *et al*. 1982; EPA 2010), but we have summarised the main guidance here.

Water samples may be taken manually or by autosamplers that can sample at fixed or variable time intervals and be triggered according to rates of water level change (Fig. 7.1). Samples gathered by hand should be taken from areas of the channel that are subject to full mixing and bottles rinsed with stream water three times before sampling to remove any residues and reduce the chances of sample contamination. Such rinsed sample bottles are acceptable for a dissolved concentration assay, but for composition analysis, acid-washed glass bottles are needed to minimise blank values.

Aqueous organic carbon includes both dissolved (DOC) and particulate (POC) forms. These are operationally separated by filtration through a 0.45 µm filter, although mesh sizes of up to 0.7 µm (GF/F) are commonly used. The sample should be filtered as soon as possible after collection as this removes particulate material and thus a large component of the bacterial community that may, through respiration, change the DOC concentration. If POC is to be assayed by loss on ignition (e.g., Waldron *et al.* 2009, Pribyl 2010), then this filter paper should be a pre-ashed glass fibre filter paper such as a GF/F. Filters should be rinsed several times with high purity deionised water to remove any blank carbon. A larger volume of sample should be collected than is needed for DOC assay to allow the filter units to be rinsed several times with the field sample.

Fig. 7.1. Field-installed autosampler with a 12V lead-acid battery for power in the plastic box, a sheathed tube to allow water to be pumped from river into 24 one litre sample bottles, all controlled by a computer linked to water pump. When the flow switch deployed in the water is triggered then sample collection commences at pre-programmed intervals until the bottles are full. Photo: Ben Smith.



There is evidence that without filtration, DOC concentration may remain unchanged for up to 90 days (Gulliver *et al.*. 2010, Murray 2012), but this will depend on the lability of the DOC pool. Thus filtration is still recommended prior to storage in refrigerated dark conditions, the latter to prevent photo-oxidation of DOC and the production of new DOC by aquatic organisms. If refrigeration is not possible then the pH of the samples can be adjusted to <2 to reduce bacterial activity (this is recommended for trace metal analysis) but care must be taken not to take the pH to 1 as at this value, humic acids can precipitate out of solution and therefore the [DOC] measured does not accurately reflect the sample concentration. If acidifying the same it is still recommended the samples are filtered first. Freezing samples has the potential to alter the DOC (via post-freezing flocculation, Gulliver *et al*. 2010, Murray, 2012) and this is not recommended at any stage prior to analysis.

* + 1. *Laboratory measurements of C concentrations*

Determination of DOC concentration is generally made using a TOC analyser. Most TOC analysers measure the CO2 produced when organic carbon in the sample is oxidized and/or when inorganic carbon is acidified (e.g., Wallage *et al.* 2006). Acidifying the sample and flushing with nitrogen or helium purges inorganic carbon from the sample, which can then be measured by a detector. For DOC analysis alone, inorganic carbon must be removed prior to analysis, and this is done by titration to pH 4 with a 0.1 M acid (the lower molarity allows more control to reaching the pH end-point) and generally with a non-chlorine based acid as chlorine can damage many TOC analysers. It should also be noted that the acid titration dilutes the sample and so the measured concentration must be adjusted for dilution by mass balance calculations using the volumes of sample diluted and water added.

The TOC is oxidised to CO2 by platinum-catalysed combustion, by heated persulphate, or with a combination UV/persulfate reactor. The CO2 is measured by a detector, usually a non-dispersive infrared (NDIR) cell, calibrated across the expected range of C concentrations.

* + 1. *Estimating DOC by spectrophotometric methods*

Direct analysis of DOC by laboratory analysis of samples is increasingly being complemented by the use of spectrophotometry (e.g., Worrall & Burt 2004). This approach relies on using the absorption characteristics of dissolved organic carbon, measured using a spectrophotometer. Spectrophotometers emit light at specific wavelengths and pass it through a sample to detect the amount of energy absorbed by the sample at that specific wavelength. This is effectively a determination of colour which is often strongly correlated with DOC concentration. Absorbance at a number of wavelengths have been used as a proxy measurement for DOC. Standard wavelengths are around 400 nm but the range used varies between 254 nm and 665 nm (see Grayson & Holden 2011 for a catalogue). After filtering, indirect measures of DOC may be made using a laboratory spectrophotometer. Samples are transferred to a cuvette which is placed in the spectrophotometer for measurement. There are a large number of commercially available spectrophotometers which are able to carry out these analyses.

This approach measures water colour and determines DOC based on the DOC vs. colour relationship. In some cases there can be a large error: up to 50% in absorbance at 400 nm and [DOC] measured on a TOC analyser (Wallage & Holden, 2010). This may be due to the presence of non-coloured DOC, or that the calibration line needs is site specific, or can demonstrate hysteresis. To significantly reduce error in calibrating colour with [DOC] it is suggested to measure absorbance at two different wavelengths on the same sample (Tipping *et al.*, 2009). A multiple wavelength approach suggests there may be further analytical insight in exploring a fuller spectrophotometric fingerprint.

* 1. *Current in-situ techniques*

In recent years there have been a number of developments in spectrophotometric technology which allow field deployment and continuous reconstruction of DOC concentrations. This technology uses absorbance at a number of wavelengths and is believed to apply corrections for particulates which may interfere with the absorbance of emitted light by the DOC. Two devices currently on the market are the Austrian S::CAN Spectrolyser[[1]](#footnote-1) and the German Trios probe[[2]](#footnote-2). These devices were originally developed for the wastewater treatment market so deployment in the field requires an external power supply. The resolution of the profiles is far more detailed than can be reconstructed with manual sampling and so offers a better understanding of C dynamics and more accurate C budget construction (Figure 7.2). However, this is relatively untested field technology with only a few research papers published (e.g., Sandford *et al.* 2010, Grayson & Holden 2012) and so we are still learning of its accuracy and precision limitations in field deployment. The following discussion focuses on the S::CAN Spectrolyser as this is the system we are more familiar with.

There is some uncertainty regarding the precise method by which DOC is calculated by these devices due to the commercial confidentiality surrounding their working. The S::CAN Spectrolyser comes with a preinstalled calibration which is made using a number of samples. An onboard (commercially secret) calibration algorithm, termed the ‘global calibration’ is used to calculate DOC concentration from a number of wavelengths and to correct for interference with suspended sediments simultaneously. The S::CAN spectrolyser terms this output DOC equivalents, DOCeq. ‘Local’ calibrations can be constructed using samples from the catchment in which the systems are deployed. This may be particularly important in the case of peatland environments where DOC may be characterised by high concentrations of humic and fulvic acids and there may be seasonal variation in the DOM composition (See section 3.3). In addition to such site-specific calibration, comparative field sample DOC concentration assay is required to assess the stability of the spectrophotometric output with the true (laboratory-measured) DOC concentration.

Fig. 7.2 Time series of stage height and DOCeq logged hourly at a small stream drainng Whitelee windfarm. During low flow there is a continuous linear decrease in DOCeq but this rises again as stage height (and so discharge) increases. For all events, peak DOCeq is logged to occur after peak event flow and at the beginning of each event DOCeq temporarily decereases. Detail like this is unlikely to be observed with manual sampling programme. Data from CLAD



* + 1. *Power*

Field measurement by the S::CAN requires power from 12V DC deep cycle batteries. Power can be conserved by using a power down function which puts the device into a sleep mode between measurements. Voltage ‘sag’ of the batteries, which can occur during cold periods, can be alleviated by using a cheap solar panel to trickle feed the battery or micro wind turbines. For field deployment, the battery and other connection adaptation equipment need to be placed in waterproof containers and kept in an area unlikely to be affected by flooding or rising waters. Wires and cables should be sheathed and buried where possible to reduce damage from grazing and burrowing animals.

* + 1. *Maintenance*

In stream deployment for long periods should consider the risk to the device of water ingress or physical damage during large flood events, dewatering of the sensor during drought and theft and vandalism.

Growth of biofilms on the spectrophotometer windows may affect light absorption and so give erroneous values. Some manufacturers offer a battery operated pneumatic cleaning system – blowing air at high pressure on to the windows to remove deposits, or use of a brush to wipe clean the window - but these have implications for battery life and may make the system more visible and vulnerable to vandalism. If proprietary cleaning devices are not used, in-situ spectrophotometers should be cleaned regularly to reduce the build-up of biofilms and filamentous algae which interfere with the measurements; exactly how often will depend on the drainage system flow (scouring) and nutrient status. Biofilms can be removed with a soft brush (a toothbrush is sufficient). Metal (manganese and iron) staining of the windows may occur in some catchments (particularly in acid waters) and this can be removed by periodic brushing with 8% HCl solution. It is currently unknown if there can be permanent staining of the windows causing an offset in the data. The time of cleaning should always be noted to later assess from the data if the DOCeq has been compromised by organic or inorganic contamination.

* 1. *Construction of C export budgets*

The total amount of carbon exported is a function of concentration and discharge, thus for determining carbon export budgets it is necessary fluvial discharge also be determined. There are a number of approaches (e.g., Walling & Webb, 1985) to estimating discharge and the one adopted will largely depend on the resources available. Here we focus on the main approaches to calculating export budgets and the significance the frequency of measurements has on the final budget. Data sets from two streams draining small catchments (< 100 ha) are used to demonstrate some basic concepts of flux calculation. However, prior to this it is important to outline why the budgets can vary depending on frequency of measurement.

Two important factors are known to affect the flux values of a stream over time and should be considered in the sampling strategy. These are (i) the effect of season (section 3.3) and (ii) the effect of flow (section 3.4 & 3.7).

The effect of season can be accounted for by sampling at the sub-monthly level. For example, nine catchments sampled at approximately three weekly intervals established a repeatable seasonal variation in DOC concentrations (Figure 3.7). The influence of event flow can be established by measuring concentrations at a range of flows (Figure 3.5). DOC flux from peat and organo-mineral soils increases with runoff (McDowell & Likens 1988, Hope *et al.* 1997) due to changes in flow pathways (section 3.2) and the greater volume of water moving through the system during events (which may be several orders of magnitude greater than the size of any dilution; Clark *et al.* 2007a). Further event flow can change characteristics over the course of a year. At some sites, it has been observed that concentrations of DOC are reduced during autumn events (Koehler *et al.* 2010) but may be greater earlier in the year (presumably a phenomenon related to the creation or supply of DOC from the peat). This suggests that DOC/flow relationships should be established from samples distributed throughout the year.

Depending on the size and complexity of the stream catchment, storm events can last for periods of time from several hours to 2 days (e.g., Clark *et al.* 2008) so sampling needs to be sufficiently frequent to capture the detail of within-event variations. There are three potential ways of determining this level of detail which will depend on the options available for measurement

1. Remaining on site and manually collecting samples for as long as possible at regular (e.g., hourly) intervals
2. Setting an autosampler which can be triggered by rising waters and takes samples at pre-determined (often hourly or two-hourly) intervals through the duration of a flood or high flow period.
3. Using an in-situ spectrophotometer logging at sub-hourly intervals
   1. *Flux calculations: case studies demonstrating the importance of length and frequency of sampling*

One data set was collected from a stream draining a moorland catchment in the Ochil Hills north of Stirling (Grieve, 1984). Samples were collected at approximately eight hour intervals from mid-1982 to mid-1983 and DOC concentration was analysed by oxidation with potassium dichromate and titration against ammonium iron sulphate (Ochil data). Further samples from the same site were collected at 4 h intervals and analysed using a Shimadzu aqueous carbon analyser in 2005. The second data set was collected from a stream draining a clear-felled forest catchment in Loch Ard Forest, west of Stirling (Grieve, 1989). Samples were collected at approximately eight hour intervals in 1988 and 1989 and DOC analysed on a TOCsin aqueous carbon analyser (Ard data). For both data sets instantaneous flow data were obtained for each sampling occasion from continuously recorded stage measurements. Whilst the difference in analytical equipment may result in changes in precision, for the illustrative examples of the sensitivity of budget calculations that follow, these changes are of little relevance.

Fig. 7.3 shows the time series of DOC and discharge from start of sampling for one event in August 2005. There is a general correspondence in their variation with time and the each major peak in discharge is matched by a peak in DOC concentration. The relationship between [DOC] and discharge for an individual event is usually non-linear but where [DOC] is regressed on log(discharge) the relationship becomes linear often with a strong correlation. This correlation between DOC and log(discharge) breaks down when examined over longer time periods, due to seasonal changes in DOC production.

Fig. 7.3. Time series of discharge (blue) and [DOC] (brown) during an August 2005 event in the Ochil Hills, Scotland. This demonstrates that both concentration and discharge can increase during event flow and therefore more C is exported in drainage, not just because of the increase in discharge. Data from Ian Grieve.



Instantaneous streamwater DOC flux at any point in time can be calculated by multiplying the DOC concentration (mg L-1) by the instantaneous discharge (L s-1) to give flux as mg s-1. However, if we have a continuous record of discharge (calculated from stream stage records), we can use equations predicting DOC from flow and temperature (or time of year) to calculate flux on an almost continuous basis. This is probably the most accurate method of calculating flux over a period of time. However we rarely have complete records for long periods of time and often estimate DOC flux by multiplying average concentrations and discharge from limited data sets.

Because DOC concentration increases with flow (section 3.4), we cannot use a simple arithmetic mean as an average in flux calculations. The best average value to use is the flow-weighted DOC mean (e.g., Schleppi *et al*. 2006). This is calculated by:

1. multiplying DOC concentration by discharge for each sample time
2. summing these values for the whole data set.
3. dividing this sum by the sum of the discharges for each sample time.

The data in Tables 7.1 and 7.2 illustrate some of the difficulties of using smaller data sets for DOC flux calculations. Table 1 shows the discharge-weighted mean DOC concentration and annual DOC flux calculated from the whole Ochil data set and from analysis of DOC in two month-long data sets sampled at daily intervals, one in early spring and one in early autumn. For these data, use of the larger mean DOC concentration sampled during the autumn months almost doubles the calculated annual flux compared to the best estimate using the full data set, while the smaller mean DOC concentration sampled in Spring reduces the calculated annual flux to less than half that value.

Table 7.1. Effect of season on annual DOC flux (kg) calculated from short-term sampling.

|  |  |  |  |
| --- | --- | --- | --- |
| **Time of sampling** | **All data** | **Spring (March)** | **Autumn (September)** |
| **Mean [DOC]** | 7.56 mg L-1 | 4.06 mg L-1 | 10.00 mg L-1 |
| **Annual flux** | 7064 kg | 2637 kg | 12157 kg |

Table 7.2 compares estimates of annual flux for the Loch Ard site based on sampling at daily, weekly and monthly intervals. Six separate subsets for each time interval were obtained by subsampling the complete data set, and discharge-weighted means and annual fluxes calculated as described above. While the mean annual DOC fluxes are very similar for the monthly and weekly sample sets, the standard deviations increase markedly with decreasing sampling frequency. Examination of the individual means suggests that the most reliable estimates of annual mean require daily sampling. The 95% confidence interval for the annual flux calculated from daily sampling is approximately 10% of mean flux whereas that for weekly sampling is more than 40% of the mean and for monthly sampling almost twice the mean. For estimation of critical losses or loads, the smallest confidence intervals are desirable. However daily sampling is most likely impractical for most field studies and this therefore demonstrate the power of in-situ loggers producing semi-continuous time series.

Table 7.2: Effect of sampling frequency on calculated annual DOC flux (kg).

|  |  |  |  |
| --- | --- | --- | --- |
| **Sampling frequency** | **Daily** | **Weekly** | **Monthly** |
| **Subset 1** | 13182 | 18478 | 4826 |
| **Subset 2** | 14259 | 12506 | 8221 |
| **Subset 3** | 15293 | 11644 | 34792 |
| **Subset 4** | 13326 | 14176 | 24519 |
| **Subset 5** | 13653 | 10281 | 6160 |
| **Subset 6** | 14231 | 12947 | 8966 |
| **Mean** | **13991** | **13339** | **14581** |
| **Standard deviation** | **779.2** | **2834.8** | **12209.0** |

* 1. *Sampling design*

There are a number of methodological approaches which may be deployed to detect and explain changes in carbon loss via aquatic pathways. The approach will depend on the length of study, timing of the initiation of the study relative to the impact of interest, resources available and logistics.

* + 1. *Over what scale to sample.*

The scale over which sampling is to be carried out needs to be considered when determining where to collect samples for [DOC] assay or deploy equipment. Water in streams integrates the signals of environmental change and processes for the catchments for which they serve, but a number of processes may obscure these signals if equipment is not sited correctly. For example, Dawson *et al.* (2009) demonstrated that changes in DOC occurring in the small peat rich catchments supplying headwater streams were not detectable in the main stem channel due to dilution from other sub-catchments and in-stream processing. To determine specific environmental changes samples should be taken from sites downstream of the area of interest but as close to the site as possible. The initial choice of sites may need to be determined by a longitudinal survey to assess the best sampling site.

* + 1. *For how long to monitor*

To detect an impact of a land use change, the ideal field set-up would deploy equipment in a controlled experimental design which allows some degree of replication (e.g., Wilson *et al.* 2010). Ideally a Before-After-Control-Impacted (BACI) methodology should be undertaken, whereby two or more catchments are monitored simultaneously before one site is impacted (preferably more than one year) and the other one(s) left unaltered as a control during subsequent monitoring (preferably for more than a year). Such a situation is rare and more likely an alternative approach is required, for example sampling above and below a site of impact or constructing a regional survey of unimpacted streams to provide an estimate of the range of values to be expected without impact for comparison to the impacted catchment.

In instances where controls are not available but samples have been collected before the impact of interest it may be possible to determine an impact based on the ‘baseline’ (e.g., Waldron *et al.* 2009). Ideally the baseline will exceed one year and encompass all seasons and flows. It is particularly important to capture samples from the uppermost 10% of flows as measured on the flow duration curve.

In the absence of baseline data an alternative approach would be to use a paired sampling methodology. This relies on the availability of two catchments with identical or similar characteristics (e.g., orientation, elevation, land-use types etc.) other than the impact of interest. In such studies changes in carbon loss in the impacted catchment relative to the non-impacted catchment may be observed (e.g., Grieve & Gilvear 2009).

* 1. *Summary comments*

This chapter outlines different approaches to calculating a catchment export budget and demonstrates how sensitive these estimates are to sampling frequency. As confidence grows in the accuracy and precision of in-situ logging of aqueous C concentrations, our budget calculations will become increasingly refined. However until then we will rely on estimates of critical losses or loads from spot sampling programmes (albeit using automated sampling equipment) and to do this we must understand fully the dynamic interaction between C concentration and flow, seasonality and impact of land use.

Appendix A: A summary of common methods of stream gauging, equipment needs and the issues associated with each technique [accurate discharge estimation is critical to accurately measuring aqueous carbon flux and requires continuous measurement of water levels (stage) to estimate total runoff]. Details on stream gauging can be found in standard hydrological text books or on the web as can equipment options.

|  |  |  |
| --- | --- | --- |
| **Stream gauging-non control methods** | **Issues/recommendations** | **Equipment needed** |
| 1. Velocity-area method: This relies on the fact that stream discharge is the product of cross-sectional area and mean water velocity. Measurement of the cross-sectional area is undertaken using manual survey techniques or in larger channels bathymetric mapping. Velocity is measured at multiple distances across the channel and either at 0.6 of water depth or at 0.2 and 0.8 of water depth and averaged. Measurement can be undertaken by wading, a cable-ways, a remote control vehicle or boat according to nature of the river. Multiple measurements across a range of discharges along with stage (water level) measurements can be used to calculate a rating curve and hence a continuous estimation of discharge if stage is logged. | * Non-control methods are not normally as accurate as control methods and large errors can occur. Best suited to short-term studies * Erosion or deposition can alter the stage-discharge relationship and this bedrock controlled reaches are best * Access can be a problem * Difficult where highly turbulent flow * Instream vegetation and sluggish flow makes measurement difficult | Propellor, electromagnetic or ultrasonic, velocity meter. Cableway, boat, or remote control boat on large rivers (in UK such sites will normally have a gauging station nearby). |
| 2. Dilution gauging: This relies on inputting in to the stream a gulp or constant injection of a tracer (e.g., salt, rhodomine, fluorescent dye) that can be detected downstream after full mixing. The level of dilution is proportional to stream discharge. Multiple measurements across a range of discharges along with stage (water level) measurements can be used to calculate a rating curve and hence a continuous estimation of discharge if stage is logged automatically. | * Difficult at high flow volumes * Thorough mixing of water required * Tracer can come out of solution and precipitate on to the bed * Input of tracer is likely to require consent. | Tracer  Constant rate injection apparatus  Water quality probe (to measure tracer concentrations) |
| **Stream gauging – control methods** | **Issues/recommendations** | **Equipment needs** |
| 1. Weirs: by ponding water weirs provide a fixed relationship between water level behind the weir and discharge. The type of weir required will depend upon channel dimensions and range of flows to be measured. The basic types are sharp or broad crested and V-notch (e.g., 90 or 120 degrees) vs. rectangular shapes. Water level can be measured manually using a stage board but normally stage will be logged automatically and using a stage-discharge equation runoff can be continuously measured. | * Control methods are highly suited to where stream gauging is going to be long term * Cost of construction * Sharp-crested weirs can be easily damaged and normally less robust than flumes * Ponding of water can causes issues in low lying areas * Licence for installation required as impacts on bedload flux and fish passsage * Structure can be drowned out in low-gradient occasions or high flows | Weir or  Flume |
| 2. Flumes: Flumes create a critical velocity in the channel by constricting flow width and as such create a constant relationship between water depth upstream and discharge. Stage discharge equations are produced as before. |

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