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Review

The effects of forest management on water quality

Nadeem W. Shah [a,*], Brenda R. Baillie [b], Kevin Bishop [c], Silvio Ferraz [d], Lars Högbom [e,f], Jami Nettles [g]

[a] Forest Research, Northern Research Station, Roslin, UK
[b] Northland Regional Council, New Zealand
[c] Swedish University of Agricultural Sciences, Uppsala, Sweden
[d] Forest Hydrology Laboratory (LHF), Forest Sciences Department, Piracicaba, SP, Brazil
[e] Skogforsk, the Forestry Research Institute of Sweden, Uppsala, Sweden
[f] Swedish University of Agricultural Sciences, Umea, Sweden
[g] Weyerhaeuser Company, USA

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ABSTRACT

Water quality is generally high in watercourses draining forested areas. However, forest management can lead to detrimental effects on water quality and the aquatic environment. Key concerns include diffuse pollution, carbon transport and harmful effects on freshwater ecology.

Here, we undertake a review of the effects of a range of forestry activities including cultivation and site preparation, fertilisation and harvesting on water quality. We attempt to summarise the literature across a wide geographical area focusing on empirical studies.

Studies report a wide range of water quality impacts after forest operations including sediment delivery, nutrient losses, carbon transport, metal and base cation releases, and changes to acidity and temperature.

Spatial and temporal resolution is an important consideration. Changes in water quality at the local scale are often not seen at the catchment level and the effects of operations may be manifest many years after the work was carried out, highlighting the importance of monitoring at an appropriate spatial and temporal scale.

The development of best management practices (BMPs) such as the use of buffers, low impact techniques and phased felling have led to significant changes in operational activity, reducing and, in some cases, preventing impacts on water quality. We highlight some of the most effective techniques that can protect water quality from cultivation, drainage, fertiliser and harvesting operations.

We also take a forward look to technological, methodological and climatic developments that may alter forest management effects on water quality.

1. Introduction

Water scarcity and contamination are the greatest pressures on global water resources, directly impacting upon our social and economic well-being and ecosystem health (UN-Water, 2021). Inappropriate land use management is contributing to these pressures and so there is a need to evaluate and improve how we manage land, including forested areas.

Forests currently cover 31 percent of the global land area, a total of 4.06 billion hectares (FAO and UNEP, 2020) and can have positive and negative effects on water quality depending on the extent and type of management activities within them. On the one hand, water draining forests is generally high quality (Kauffman and Belden, 2010) and afforestation can have positive effects on water quality (Duffy et al., 2020). This is partly due to the protective function of forests and many of the world's largest cities rely on water draining forest protected areas (Dudley and Stolten, 2003; Liu et al., 2021; Motta and Haudemand, 2000). On the other hand, forest management and operations have the potential to impair water quality by causing, for example, diffuse (non-point source) pollution through nutrient runoff following fertiliser application, or increased sedimentation and carbon transport following cultivation and harvesting (Laudon et al., 2009; Nisbet, 2001).

Dissolved organic carbon (DOC) transport is important in many

* Corresponding author.
E-mail addresses: nadeem.shah@forestresearch.gov.uk (N.W. Shah), brendab@nrc.govt.nz (B.R. Baillie), kevin.bishop@ma.slu.se (K. Bishop), silvio.ferraz@usp.br (S. Ferraz), lars.hogbom@skogforsk.se (L. Högbom), jami.nettles@weyerhaeuser.com (J. Nettles).

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contexts including global warming related terrestrial carbon loss (Waldron et al., 2019) and drinking water quality treatment (Valdivia-Garcia et al., 2019). Concern has grown because increasing DOC concentrations have been reported in forested and non-forested environments (Burns et al., 2006; Evans et al., 2006).

Another significant water quality issue is soil and surface water acidification, which, although declining in North America and Europe (Garmo et al., 2020; Webster et al., 2021), is still relevant particularly in forested areas with sensitive geology and high atmospheric deposition of pollutant sulphur and nitrogen (Zhu et al., 2016). The mobilization and methylation of mercury after forest harvest has also been identified as an issue due to the chemical's toxicity and potential effects on freshwater ecology (Bishop et al., 2020).

Many species, including fish and freshwater invertebrates, are sensitive to changes in background water quality (Osterling et al., 2010; Shaw and Richardson, 2001; Vuorinen et al., 1998; Wood and Armitage, 1997), which has led to increasing scrutiny of the effects of forest management on the water environment. The concern is justified due to the abundance of species living in natural (Vie et al., 2009) and plantation forests (Brockerhoff et al., 2008).

With the additional impacts of climatic extremes on water quality and global water resources (UNESCO, UN-Water, 2020), it is essential that land management does not add to the pressures on the water environment, but instead helps to reduce and even enhance water quality (FAO et al., 2021).

The need for sensitive forest management to protect water resources has been recognised and is reflected in the increasing number of studies on forestry and water quality, particularly since the 1990s until which time most forest hydrology studies had focused on the issue of water quantity in relation to drinking water supplies and ecological flows (McCulloch and Robinson, 1993). These water quality studies have naturally led to an increase in published and grey literature with useful reviews appearing over time: Shepard (1994) reviewed the effects of forest management on water quality in wetland forests in the USA, Schoenholtz (2004) provided a useful summary of the effects of forest management on water quality and Koralay and Kara (2018) focused on the effects of harvesting on water quality. National research and forest agencies have provided region-specific data, guidance and review (Brown and Binkley, 1994; Fulton and West, 2002; Hornung and Adamson, 1991; Neal et al., 1992; Newson, 1990; Pike et al., 2010).

However, as far as the authors are aware there has been no global review that covers the effects on water quality of a range of forestry activities including cultivation and site preparation, fertilisation and harvesting; under these sub-headings, we attempt to summarise the literature on forestry and water quality across a wide geographical area. The review focuses on key chemical and physical water quality parameters including nutrients, carbon, water colour, turbidity, suspended solids (SS) and acidity.

Due to the vast amount of literature published across multiple disciplines we narrowed the scope to focus on empirical rather than modelling studies and those that reported on streamwater quality rather than soil water; this allowed us to concentrate on the observed effects of forestry on downstream water quality. It was also unfeasible to include the impacts of road construction and pesticide applications. We recognise that water quality is integrally linked to hydrological flows, but it was outside the scope of this paper to review the effects of forest management on flow hydrology.

With the backdrop of climate change and global warming it is essential to recognise that water quality is subject to natural variations and that these need to be differentiated from land use effects including the effects of forest operations; this is even more important during and after extreme events such as droughts and storms and so we make some attempt to include studies that try to disentangle the effects of climate from land use.

Table 1
Soil cultivation techniques used in forestry (Carling et al., 2001; Forestry Commission, 2019; Paterson and Mason, 1999).

Technique	Description
Direct planting/no cultivation	Planting trees directly into cuts in the soil made by a spade.
Moling	Creating a subsurface drainage channel with a mole plough; a pointed cylinder on the lower edge of a bar is passed through the subsoil, usually at 25–45 cm depth. Used for draining heavy soils of almost uniform slope. The addition of 'wings' to the mole can fracture compacted layers.
Mounding	Formation of a small mound of soil, usually 20–30 cm in height, on which to plant a tree. Can be formed by excavators or continuous mounding machines. Excavator mounding can form three different types of mound: inverted mounds place the soil back in the hole; hinge mounds flip the soil over leaving one edge of the upper surface intact; trench mounds are created from soil excavated from trenches (which may be filled in afterwards).
Ploughing	Cultivation of soil in continuous ridges and furrows.
Ripping (deep subsoiling)	Deep cultivation to depths of 30–105 cm for the purpose of shattering compaction or induration; or disrupting deep-lying iron pans or cementation.
Scarification	Shallow continuous or discontinuous (patch scarification) cultivation designed to create suitable positions for tree planting or a seed bed for natural regeneration. Breaking up surface litter, humus and vegetation to expose mineral soil to a shallow depth, usually < 15 cm
Screefing	Very shallow (usually < 10 cm) form of cultivation involving the removal of herbaceous vegetation and soil organic matter to expose patches of bare soil for planting.
Subsoiling	Cultivation to 40–50 cm depth to disrupt rooting obstructions such as iron pans. The function is to reduce bulk density within normal rooting depth; to disrupt iron pan formations at intermediate depth (10–35 cm); or to provide seepage channels at 35–50 cm in stony gley soils outside the scope of moling.
Furrowing; V-blading	Continuous formation of shallow V-shaped furrows to 15–25 cm depth by a blade (V-blade, divider or shallow plough).

2. Forest management

2.1. Drainage, cultivation and planting

Cultivation is carried out to create favourable growing conditions for tree planting; the aim is to loosen compacted soil, reduce weed competition and create a raised planting position; this last point is particularly important on wet soils as it helps reduce the soil moisture content and increase oxygen levels for improved root development (Paterson and Mason, 1999). It also reduces the risk of frost damage to young trees. Drainage is carried out, often together with cultivation, to remove water on sites where tree growth is inhibited by high soil moisture levels. Drainage can increase water flow and erosion and lead to water quality deterioration (Finér et al., 2021).

There are a variety of soil cultivation/drainage practices with different levels of soil disturbance including scarification, sub-surface treatments, mounding and ploughing (Table 1). Scarification prepares planting positions by scraping off surface vegetation and redistributing brash; sub-surface treatments include moling (inserting continuous subsurface channels for drainage) and sub-soiling (breaking the soil structure without any mixing of the horizons); mounding provides regularly spaced heaps of soil for direct planting; ploughing is the most intensive cultivation technique and is used to form continuous ridges and furrows (Paterson and Mason, 1999). Harrowing may also be used where the organic layer is removed from furrows (width ca. 0.50 m) and turned over to form ridges consisting of a double organic layer plus dead ground vegetation and logging residues (Piirainen et al., 2007). Cultivation techniques may involve the use of large machines which in themselves can lead to considerable soil disturbance and compaction particularly on sensitive soils and under adverse weather conditions.

Planting presents less of an issue than cultivation particularly where planting is carried out by hand. Where machine planting is undertaken, there is potential to disturb soil particularly in newly cultivated areas.

Established in the 1960s, two of the longest running forest hydrology studies in the UK, at Coalburn and Plynlimon, have provided useful data on the effects of forest management on water quality. At Coalburn in Northern England, on predominantly blanket peat, pre-drainage suspended sediment concentrations were < 4 mg L^{-1} but during ploughing average suspended sediment concentrations increased to 30 mg L^{-1} in dry periods and 150 mg L^{-1} in rainy periods; higher concentrations were recorded when blocked plough drains were cleared by hand with max recorded values of > 7000 mg L^{-1} and average values of 300–1700 mg L^{-1} (Robinson et al., 1998). No increase in sediment was recorded during tree planting highlighting that drainage was responsible for the high level of sediment transport (Robinson et al., 1998). The authors also found that phosphate levels increased up to 2.1 mg L^{-1} after ploughing operations, from background annual concentrations < 1 mg L^{-1}, with the relatively high concentrations mainly due to fertilisation of the soils prior to ploughing (Robinson et al., 1998). On blanket peat at Llanbrynmair, Wales, in one catchment total sediment loads increased from 37 to 90 kg ha^{-1} yr^{-1} after ploughing and in another from 7 to 31 kg ha^{-1} yr^{-1}; the authors thought it likely that an unploughed strip adjacent to stream courses reduced and delayed sediment transport between the furrow and the stream (Francis and Taylor, 1989), a relatively early indication of the benefit of Best Managment Practices (BMPs) in protecting water quality.

In contrast to these studies, at Argyll in Scotland water quality was not impacted after extensive ploughing and drainage of peaty soils (Nisbet et al., 2002) and at Balquhidder, also in Scotland, no changes in water quality were attributed to ploughing and planting (Harriman and Miller, 1994). At the latter site, the largest sediment loadings were associated with climate rather than forestry, highlighting the importance of differentiating the effects of climate from land use management in environmental studies; at Argyll the results were attributed to good forestry practice (Nisbet et al., 2002) (see section 2.1.3 below). More recent data, also from Scotland, showed that stream turbidity levels, colour, acidity and DOC were unaffected by ploughing and drainage due to the use of good forestry practice measures that included the use of wide (50–70 m) riparian buffer areas; annual mean NO_3-N (<0.15 mg L^{-1}) and NH_4-N (<0.11 mg L^{-1}) concentrations remained low throughout the monitoring period revealing no increases in response to pre-planting ploughing and drainage (Shah et al., 2021).

Extensive drainage and drain maintenance has been carried out for forestry in Fennoscandia on both mineral and peat soils (Paavilainen and Paivanen, 1995), resulting in not only increased transport of sediment and organic matter to receiving waters (Heikurainen et al., 1978; Joensuu et al., 1999; Finer et al., 2021) but also nutrients (Finer et al., 2021; Lundin and Bergquist, 1990; Marttila et al., 2018) and metals (Estlander et al., 2021; Lundin and Bergquist, 1990). A wide range of suspended sediment concentrations have been reported post-drainage with annual means typically < 20 mg L^{-1} (Kenttamies, 1981; Marttila et al., 2018) although during high flow after rainfall or snowmelt concentrations as high as 408 mg L^{-1} were recorded (Marttila et al., 2018).

On mineral soils, site preparation by harrowing on a clearfelled site led to increased DOC, N and P in soil water compared to pre-treatment levels with increased levels persisting for 1–2 years for inorganic N and P, and 5 years for DOC and organic N; due to the rapid recovery of ground vegetation and low N deposition loads, the leached amounts remained small (Piirainen et al., 2007).

In Finland, ditch maintenance increased suspended solids in runoff water with concentrations averaging 4–5 mg L^{-1} in controls and pre-maintenance, but increasing to 45.8 mg L^{-1} post-maintenance. The size of the increase was dependent on the area subjected to ditch maintenance and the dominant soil type at the bottom of the ditches (Joensuu et al., 1999). These findings were supported by Nieminen (2003) who found on a drained, nutrient poor, Scots pine mire that site preparation by ditch mounding led to increased transport of suspended solids from one of the four monitored areas, namely where the ditches reached the mineral soil under the peat layer.

A review of 23 studies (22 from Finland and 1 from Sweden) in the Boreal zone, (Nieminen et al., 2018a, 2018b) indicated that ditch maintenance increased erosion and exports of suspended solids and particulate N and P; however, impacts were minor for dissolved N (mean NH_4-N < 0.12 mg L^{-1} and NO_3-N < 0.9 mg L^{-1}) and P (Total Dissolved P < 0.10 mg L^{-1}), and DOC exports decreased rather than increased (Nieminen et al., 2018a, 2018b).

Reduced carbon transport was also found following drainage on peatland (Lundin and Bergquist, 1990; Nieminen et al., 2018a, 2018b), where lowering of the water table led to reduced contact of water with organic-rich peat soils (Lundin and Bergquist, 1990) and increased adsorption of negatively charged organic molecules, thereby decreasing DOC concentrations in soil water (Nieminen et al., 2018a, 2018b). Organic N and PO_4-P concentrations also decreased after drainage and lowering of the water table, the former due to the increased influence of mineral rather than bog groundwater on streamwater chemistry; the P decrease may have been due to increased sorption to the peat soil (due to increased oxygenation), precipitation to stream sediments and biological uptake (Lundin and Bergquist, 1990). In contrast, Nieminen et al. (2017, 2018b) and Finér et al. (2021) indicate that nutrient and carbon transport from drained peatlands may be significant due to the legacy effect of past first-time drainage. Nieminen et al. (2017) found that Total N and Total P concentrations increased with years since drainage, but recognised that other factors such as geographical location, weather conditions, site characteristics, catchment area, historical forest operations such as fertilisation and soil mineralisation processes may also contribute to higher nutrient concentrations in older drainage areas (Nieminen et al., 2018b). The authors acknowledged that the geographical extent of the studies is rather limited, including only Finnish studies (Nieminen et al., 2018b).

In the US, water quality studies have often assessed the effects of site preparation and harvesting together. Blackburn et. al. (1986) found that shearing, windrowing and burning treatment led to sediment concentrations averaging 2119 mg L^{-1} the first year after harvest, whilst chopped and burned treatments yielded sediment concentrations similar to the controls. In 2002, the same sites were harvested and prepared using BMPs with site preparation treatments of herbicide only, subsoiling, fertilisation and herbicide, or unharvested control. While sediment losses were elevated with site preparation, the maximum was still one-fifth that of the 1981 pre-BMP treatment and not significant on the large herbicide only watershed (McBroom et al., 2008a). Beasley et al., (1986) compared a herbicide only treatment with more intense site preparation, namely shearing and windrowing. In the first year following treatment, no significant sediment losses were found in the herbicide only treatment, but significant increases were found in the sheared and windrowed treatment compared to the herbicide only and control treatments.

Lakel et al. (2010) evaluated 16 sites in Virginia, USA, that were harvested, burned and hand planted. They looked at sediment trap data to estimate USLE (universal soil loss equation) annual erosion rates of between 7.1 and 15.6 tonnes ha^{-1} yr^{-1} but concluded that riparian buffers trapped 97% of eroded sediment before it reached a stream. Wynn et al., (2000) also looked at clearfelling and site preparation in Virginia, USA, but separated out the effects of harvest from site preparation. Two sites were harvested, one with BMPs and one without, and these were compared to a control site. BMPs included a streamside management zone with a minimum of 15.2 m width delineated on each side of the stream where minimal harvesting was conducted, water bars installed on primary skid trails to divert surface flow to an area of undisturbed litter, and landings seeded with grass to establish ground cover until the time of replanting. The average sediment yield from the

BMP site did not change significantly pre and post treatment, while the average annual sediment yield from no-BMP site was an order of magnitude above the control and BMP sites.

In North Carolina, USA, a long-term study was established to measure the effects of drainage and silviculture on water quality (Amatya and Skaggs, 2011). Amongst the findings were that sediment and most nutrient concentrations in runoff were elevated after harvesting on the drained sites (Amatya et al., 2006). Controlled drainage through the addition of seasonally adjusted weirs was found to reduce nutrient and sediment exports, predominantly by reducing outflow (Amatya et al., 1998). Lebo and Herrmann, (1998) measured water quality in drainage ditches in North Carolina, USA, following harvest. They found minimal change in Total Suspended Solids (TSS) and small elevations in N and P in runoff, from 2.1 to 2.2 kg ha^{-1} yr^{-1}, and 0.12 to 0.36 kg ha^{-1} yr^{-1}, respectively.

On peatlands in Quebec, Canada, Prévost et al. (1999) found that summer flows increased after drainage, which led to increased suspended sediment, conductivity and nutrient export (NH_4 NO_x); Ca, Mg, Na and S also increased. However, conductivity and the increased nutrient concentrations remained within the acceptable criteria for aquatic organisms. After drainage, pH also increased by one unit, a result attributed to increased runoff from the upland part of the treated watershed.

In Brazil, sediment delivery increased after the preparation of soils for planting Eucalyptus, and contour planting appeared to lower soil loss rates (da Silva et al., 2011). Site preparation by minimal cultivation (ripper, pit digger and hand cultivation) reduced erosion rates and nutrient losses, contributing to reduced sediment input to streams in the management areas (Goncalves et al., 2008). We found no data on the effects of cultivation for forestry in Africa.

2.1.1. Fire as a land preparation

Effects on soil and hydrology are also found after prescribed fire, which is used as a site preparation method in some areas including Africa (Savadogo et al., 2007), Australia (Klimas et al., 2020), southern Europe (Fernandes et al., 2013) and North America (Ryan et al., 2013). Fire is rarely used in New Zealand due to concerns over the loss of organic matter and the potential for fire, and so its use is primarily confined to burning excessive accumulations of logging residues.

In Australia's fire-adapted forests, prescribed fire is used to prepare sites for eucalyptus seeding and is also widely used in the beginning of the dry season to reduce under-story fuel loads and associated wildfire risks, particularly in dry eucalypt forests (Klimas et al. 2020). Due to the high clay content in forest soils, high intensity burns make them prone to water repellency and there is a risk of nutrient depletion in nutrient-poor soils.

Smith et al. (2010), assessed the effects of prescribed fire on suspended sediment and nutrients in two small Eucalyptus catchments (133 ha and 87 ha) in south-eastern Australia. Suspended sediment and phosphorus yields peaked at 11.5 kg ha^{-1} yr^{-1} and 0.016 kg ha^{-1} $year^{-1}$ respectively. A repeat burn in one catchment in 2006 led to lower sediment and nutrient export compared with 2005 due to the low rainfall in that year, underscoring the important role that climate plays in water quality fluctuations. The overall effect of the burns on suspended sediment and nutrients was minor, and water quality recovered within 12–18 months in line with the recovery of vegetation cover.

Time of year can also influence the severity of burning and subsequent effects on water quality. Townsend and Douglas (2000) compared early dry season and late dry season prescribed burns and no burning on water quality over a 3-year period in tropical northern Australia. The forests were predominantly *Eucalyptus* with an understory of tall C4 grasses. For the late season burn, most of the post-burn sediment and nutrients (Volatile Suspended Solids, TSS, P, N, Fe and Mn) were exported in high concentrations during episodic run-off events and were up to 10 times higher than concentrations measured in the following wet season. In comparison, the early low intensity burn had minimal effects on water quality with concentrations similar to that in the unburnt control catchment. Regardless of burn type, the overall impact on water quality was low, and attributed to low slopes, low soil fertility, a protective gravel surface, and length of time between burning and the first run-off event. However, the lack of pre-burn data limited the significance of these results.

In South East Asia there is widespread use of fire to clear previously logged forest and other degraded land in preparation for oil palm, rubber, or pulpwood plantations. An experiment in the Mendalong research area (Sabah, Malaysia) compared the effects of different logging and burning treatments on water quality (Malmer 1996). Stormflow SS concentrations increased at all sites after burning and felling, but concentration increases were lower with light selective logging and manual extraction. Stormflow dissolved nutrient concentrations (total-N, total-P, K, Ca, Mg) increased following clearfelling and burning from 10 to 100 times compared with baseflow concentrations. The lowest nutrient increases were associated with selective logging and manual extraction. This study highlighted the risk of high sediment and nutrient losses in a humid tropical environment where disturbance is high and storm events initiate run-off. Burning activated large losses of nutrients regardless of the degree of soil disturbance (Malmer 1996).

There are a few studies on soil solution chemistry after prescribed burning following final felling in Fennoscandia. Ring et al. (2013) initiated a study to examine how burning of a clear-felled area affected the soil and soil-solution chemistry in boreal Sweden. Soil-solution NO_3-N concentration in the burnt area peaked at 0.50 mg L^{-1} and the mean concentration during the first seven seasons was 0.13 mg L^{-1}. In the unburnt area, the NO_3-N concentration peaked at 3.1 mg L^{-1} and the corresponding mean concentration was 1.0 mg L^{-1}. Although the general level of NO_3-N was low in this study, burning largely counteracted the N concentration increases that can follow final felling.

In a recent summary of the effects of prescribed fire in the Eastern USA, Hahn et al. (2019) presented a conceptual model predicting water quality outcomes of fire based on intensity, severity, and the resulting effects on soil and vegetation. They found prescribed fire had little negative effect on water quality, and over time may shift vegetation and forest floor composition to the benefit of water quality and yield. Neary (2019) also indicated that prescribed fire had little effect on water resources and lowered the risk of wildfire, which has a much greater potential for affecting peak flow and sedimentation.

A review of prescribed fire and water quality research that included North America, Australia and Southern Europe concluded that while prescribed fire can lead to increased sediment and nutrient transport, the increases were lower than those caused by other silvicultural practices and would cause little environmental impact; moreover, increases were far lower than those found after wildfire (Klimas et al., 2020).

2.1.2. Summary of cultivation effects

Across all geographical regions, studies indicate that the main impact of cultivation for forestry is sediment transport with the highest concentrations usually associated with high flow events. Studies indicate that there may be long-term effects of drainage on water quality although data is limited to Fennoscandia. Nitrogen concentrations generally remained low after cultivation in Europe, particularly on soils with high organic matter such as peats; this is most likely due to the relatively low availability of nitrogen in the peaty soils and increased nitrogen demand by the growing forest and other vegetation (Aber et al., 1998; Nisbet and Evans, 2014). In North America, NO_3-N releases were reported but usually below 10 mg L^{-1}, a commonly used standard used to protect surface water ecology. Phosphate increases after cultivation are reported across all areas but appear to be greater in the UK, Fennoscandia and S.E. Asia than North America and Australasia. This is perhaps related to the prevalence of organic matter soils, which are not as efficient at adsorbing phosphorus as mineral soils.

In general, prescribed fire tends to have lower fuel loads, burn severity and intensity than wildfires, and their impact on water quality is

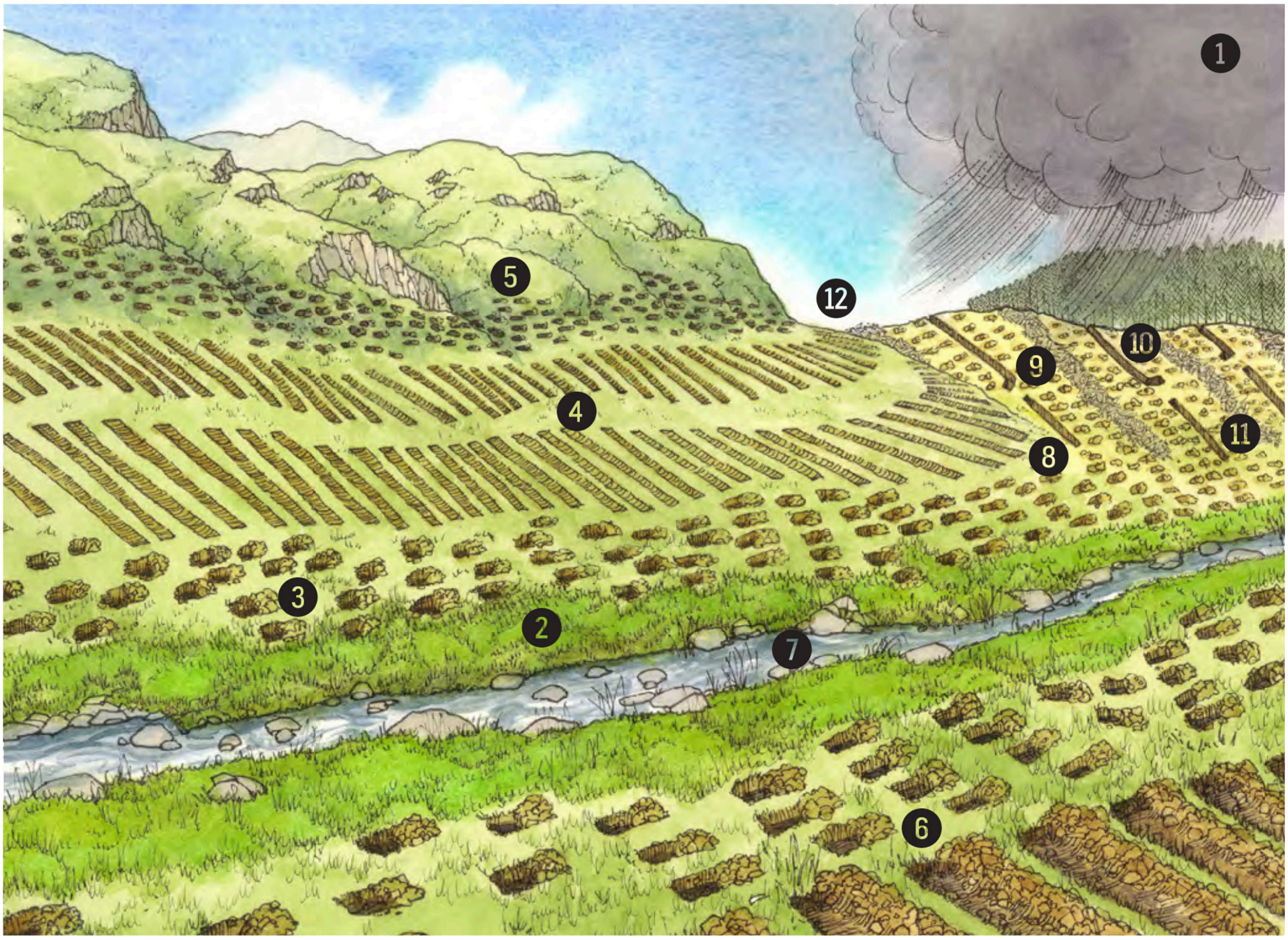

Fig. 1. Best Management Practices to protect water quality from cultivation operations. 1. Considering the weather and carrying out cultivation operations during dry periods whenever possible. Determining the appropriate cultivation method for the site conditions to minimise soil disturbance, erosion and sediment delivery. 2. Avoiding ground close to watercourses, springs, wells or boreholes; keeping buffers and Riparian Management Zone (RMZ) as wide as practicable. 3. Avoiding cultivation within buffer/RMZ areas or, if necessary, using low impact techniques such as hinge or inverted mounding. 4. Leaving breaks in plough lines (and any associated subsoiling) at regular intervals; following contour lines to avoid erosion when subsoiling. 5. Using only discontinuous forms of cultivation on steep slopes. 6. Restricting the depth of ploughing to reduce soil disturbance. 7. Avoiding fording of streams and rivers; crossing at right angles at a point where the stream bed is straight and uniform. 8. Not digging spoil trenches that can discharge directly into watercourses. 9. Orientating spoil trenches so that they cannot intercept or carry large volumes of water; turning out the bottom 2 m length of each trench to alternate sides to dissipate flows. 10. Not filling trenches created for mounding with fresh brash/slash. 11. Restricting the length of trenches to <30 m, or fully integrating trenches into the drainage system; restricting the gradient to 2° (3.5%). 12. Installing drains at the same time or immediately after cultivation operations. *Image reproduced from* Forestry Commission (2019) *© Crown Copyright.*

relatively minor in comparison to wildfires. The greatest risk is increased erosion and sedimentation (and contaminants associated with sediment), and run-off in high rainfall events in the immediate post-fire period. However, sediment loads are usually lower than found with other silvicultural techniques.

2.1.3. BMP for cultivation and drainage

The development of best practice guidance has led to significant changes to cultivation practices. This in turn has reduced and, in some cases, prevented impacts of cultivation on water quality where best management practices (BMPs) are employed. Fig. 1 and Fig. 2 illustrate BMPs that can help reduce and prevent water quality impacts from cultivation and drainage, respectively.

Studies have shown that impacts on water quality can be avoided by shallow ploughing and using furrow-end buffer strips on steeper (>5°) slopes (Nisbet et al., 2002). Leaving uncultivated buffers between furrows and the stream can reduce and delay sediment transport to watercourses (Francis and Taylor, 1989). Limiting ploughing to gentle to moderate slopes, and the use of 5 m furrow end buffers and wide (50–70 m) riparian buffer areas are also effective techniques that can protect water quality (Shah et al., 2021). Lakel et al. (2010) assessed sediment retention by buffers of varying width (7.6 m to 30.4 m) and found that sediment retention was effective regardless of the buffer width.

In forest plantations in Brazil, keeping litter and crop residues in the soil, followed by soil preparation in planting rows or holes, has helped reduce erosion and promoted nutrient retention in soils (Goncalves et al., 2002).

Ensuring that ditch maintenance on peaty soils does not reach deeper mineral soils can mitigate suspended solids and particulate nutrient transport (Nieminen et al., 2018a, 2018b), and leaving part of the soil surface intact can minimise leaching (Piirainen et al. 2007). Sediment losses can be significantly reduced by minimising the number and size of skid trails and ensuring ripping is carried out on contours because rips can serve as preferential flow paths that carry sediment to streams (McBroom et al., 2008a).

2.2. Fertilisation

Fertilisers are used in managed forests to address nutrient deficiencies and to increase forest productivity, but they can have an adverse impact on the water environment if nutrient runoff (Bergh et al., 2008; Binkley et al., 1999) enriches local watercourses (Smith et al.,

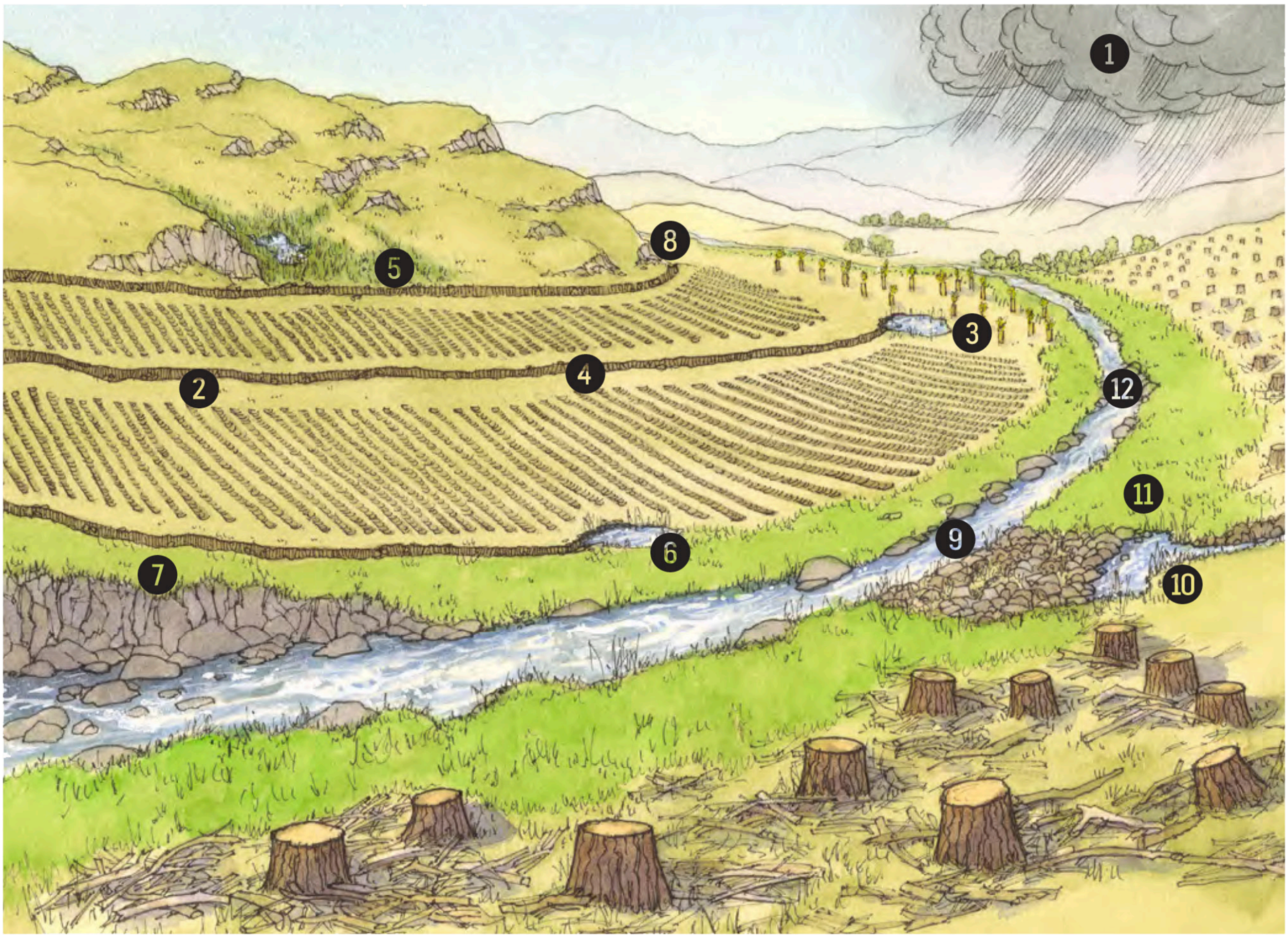

Fig. 2. Best Management Practices to protect water quality from drainage operations. 1. Considering the weather and aiming to carry out drainage works (including drain maintenance and silt trap cleaning) during dry periods whenever possible. 2. Cutting drains to run at an even gradient of 2° (3.5%) or less leading towards the head of the valley; ensuring water does not discharge into lower cultivation channels. 3. Ending drains in a shallow turnout. 4. Spacing drains so that the volume of run-off does not exceed the capacity of the drainage system. 5. Providing 'cut-off' drains so that plough furrows do not carry significant volumes of water from wet areas above. 6. Stopping drains at the edge of buffer/RMZ areas, preferably on flat ground where water can fan out. 7. Ensuring drains do not discharge to the edges of steep gully sides or unstable slopes. 8. Avoiding drains diverting water to adjacent catchments. 9. Not ending drains in natural channels, ephemeral streams or old agricultural drains. 10. Redesigning existing drainage systems to meet current standards and correcting any erosion problems; ensuring restocking drains discharge to a minimum 10 m wide buffer area. 11. Where an existing drain has become a sizable and stable watercourse, treating it as a natural watercourse and establishing buffer/RMZ areas along its length. 12. Avoiding fording streams and rivers. *Image reproduced from* Forestry Commission (2019) © *Crown Copyright.*

1999). This nutrient transport, a form of diffuse pollution, can in extreme cases result in eutrophication of receiving waters, where algal growth depletes oxygen levels eventually leading to adverse impacts on aquatic life (Smith, 2003). Relatively small rises in phosphate concentrations can cause unwelcome ecological changes that disturb the ecosystem balance, especially within oligotrophic (nutrient-poor) standing waters such as lakes and reservoirs. Therefore, considerable efforts have been made to minimise the impact of forest fertilisation on the water environment (Hensley et al., 2002).

In some parts of the world forest fertilisation has declined greatly in recent years (Albaugh et al., 2019; Lindkvist et al., 2011) for several reasons including environmental protection, cost, and reduced need on initially fertilised areas where nutrients will be provided by tree debris in subsequent rotations. However, significant forest fertilisation does still take place (Smethurst, 2010) and has increased in some areas after a decrease (Lindkvist et al., 2011), partly driven by the desire to increase economic production, enhance carbon sequestration and carbon stocks, and expand production of forest biomass as an alternative to fossil fuels (Gallo et al., 2021; Hedwall et al., 2014). Therefore, forest fertiliser applications still pose a risk to the water environment.

Fertiliser losses have been shown to be greatest during the first six months after application (Beltran et al., 2010; Binkley et al., 1999; Liechty et al., 2006; Shah et al., 2021) but can be elevated for three years (Harriman, 1978), and even 5 to 10 years on drained peatland (Kenttamies, 1981). Peak concentrations are often seen in the period immediately after fertilisation where there is potential for most runoff. A review of 14 studies in N. America indicated that peak NO_3-N concentrations usually occurred in the first two months after fertilisation and ranged from 0.1 to 4 mg L^{-1} (Brown and Binkley, 1994). On peat in England, phosphate concentrations increased from 0.01 mg L^{-1} to 0.27 mg L^{-1} soon after fertilisation with concentrations in succeeding storms as high as 1.5 mg L^{-1} (Robinson et al., 1998). Concentrations were lower at a site in Scotland, with Total P peaks of 0.03 mg L^{-1} recorded during the time of fertilisation, although a peak of 0.11 mg L^{-1} was seen over one year later, most likely due to mobilisation of soil adsorbed phosphorus (Nisbet et al., 2002), or perhaps the delayed dissolution of rock phosphate (Nieminen and Jarva, 2000). Similar concentrations were found at another site in Scotland, where concentrations after repeated fertiliser applications remained below 0.1 mg L^{-1}; larger peaks in concentration of 0.1 to 0.16 mg L^{-1} were recorded outside of fertilisation periods and appeared to be unrelated to the applications (Shah et al., 2021). Hensley et al. (2020) reported negligible effects on nutrient export and stream biota following forest fertilisation despite adding four times the amount recommended by Florida's BMPs (280 kg N ha^{-1} and

90 kg P ha^{-1} in any 3-year period).

The method of application is an important factor, with greater losses to surface waters from aerial compared to ground based or hand applications. Improvements to fertiliser practice including better helicopter targeting systems and the use of buffer areas have succeeded in reducing phosphate losses to water (Binkley et al., 1999; Nisbet, 2001; Nisbet et al., 2002), although phosphate peaks caused by direct wash-off or aerial drift of fertiliser into the stream have been reported (Nisbet et al., 2002). Uncertainties remain over fertiliser applications to particular soil types such as deep peat, which is less able to absorb and retain applied P and presents a greater risk of nutrient runoff to local streams. The use of iron enriched P fertilisers to improve P adsorption to peat soils and reduce leaching was assessed in Finland; the authors concluded that the risk of fertiliser P leaching to water bodies from peat soil is low for Fe enriched P fertilisers (Nieminen et al., 2011).

Fertilisation can be a significant issue in areas with high acid deposition where the combined load from atmospheric deposition and fertilisation can lead to nitrogen leaching to surface waters (Gundersen et al., 2006). Matzner et al. (1983) working on mineral soils in Germany reported increased soil acidity following N-K fertilisation due to exchange of Al ions; they also reported higher concentrations of Al and Mn in seepage water after fertilisation and increased rates of NO_3-N loss after liming.

In the Boreal region nitrogen is the mineral nutrient that limits plant growth and so forest fertilisation has been used to increase biomass and value (Nohrstedt, 2001). However, there are strict regulations concerning N-fertilisation based on site quality and species composition. The standard dose is 150 kg N ha^{-1} given in a single application 10 years before final felling. As a rule of thumb about 80% of the added N ends in the soil, mainly in the humus layer; the highest recorded direct losses to soil solution were 8% (equal to 12 kg N) of the added N (Nohrstedt, 2001). This additional storage of N in the soil raised questions about nitrogen leaching following final felling. Studies on the effects of final felling at old fertiliser experiment sites indicated that previous N fertilisation of N-limited forests does not affect the soil-solution chemistry significantly after whole-tree harvesting (Ring et al. 2012; Ring et al. 2018); they also reported short-term effects of N fertilisation with 150 kg N ha^{-1} on soil–water chemistry and no long-term effects. The outcome depends largely on the previous fertiliser dose, the time since last fertilisation and the original site productivity. It seems that the C to N ratio in the humus layer to some extent could explain why fertile, N rich sites have a more pronounced N loss (Andersson et al. 2002).

Lundin and Nilsson (2014, 2021) assessed the effects N fertilisation (150 kg N ha^{-1} and also Ca, Mg and B) by tractor in a 45 ha Scots pine dominated catchment. In the first year after treatment, the streamwater nitrate concentration increased from 0.05 mg L^{-1} to 3.3 mg L^{-1} on average. Other elements showing increased concentrations were ammonium (300%), boron (threefold), magnesium (80%), calcium (60%), potassium (50%) and sodium (40%); pH decreased in the first six months by 0.2 pH units.

Recycled wood-ash has been used as a fertiliser treatment to counteract the increased loss of nutrients from soils following whole tree harvesting (WTH) (Nohrstedt, 2000). Wood-ash contains potassium and phosphorus, but little if any nitrogen. In southern Sweden wood ash was applied to a Scots pine stand on a drained peatland and soon after the application concentrations of boron, calcium, potassium, lithium, magnesium, manganese and sulphate in the ditch water, and the electrical conductivity all increased (Ring et al., 2011). Some variables showed elevated concentrations for a few months only, whilst others were elevated for up to at least three years. A long-term increase for Total P was found and although peaks were also seen for NH_4-N and PO_4-P, they were at levels found in the pre-application reference period (Ring et al., 2011). Application of wood ash to drained boreal peatlands resulted in elevated concentrations of S, K, Na, Cl and Mg in runoff water, relative to the control, even after 10–11 years; however, no increased leaching of P, N, DOC or heavy metals was detected (Piirainen et al., 2013).

Nieminen et al. (2007) suggested that the low leaching of P following fertilisation with wood ash on drained peatlands is due to P being adsorbed by Al and Fe during weathering of the ash fertilisers. However, they could not say whether the adsorption of P occurs with the Al and Fe present in the ash or the native Al and Fe compounds present in soil before ash fertilisation.

Eucalyptus plantations in Brazil make highly efficient use of nutrients from fertilisation (Stape et al., 2004). However, in catchments with Eucalyptus plantations, Ranzini and Lima (2002) observed that some nutrient losses were associated with fertilisation. Overall, the effects of Eucalyptus plantations on subsurface drainage and groundwater contamination are low due to the efficient use of nutrients, limiting concentration increases even after disturbances such as forest harvesting (Laclau et al., 2010).

On poor fertility soils in the Brazilian Savannah (Cerrado) (Yamada, 2005), a paired catchment study compared the effects of sugar cane, Eucalyptus plantation and natural vegetation on water quality. Results showed that conversion of the Cerrado to Eucalyptus led to increased concentrations of cations and organic carbon in small streams, but levels were lower than in the sugar cane plantations leading the authors to conclude that Eucalyptus plantations exerted a moderate impact on the streamwater (Silva et al., 2007).

In Africa, a study in conducted in Kenya reported higher NO_3-N concentrations from streams in tree (and tea) plantation compared to natural forest, with fertilisation indicated as the source of the NO_3-N (Jacobs et al., 2018).

Several fertiliser studies were undertaken across New Zealand's managed forests in the 1970s where either urea or superphosphate was aerially applied to nutrient deficient *Pinus radiata* stands (Neary and Leonard, 1978). Not all N and P parameters were measured in these studies, but across these studies, Total-N, Organic-N, Ammoniacal-N, Nitrate-N, PO_4-P and Total P peaked at 0.79 mg L^{-1}, 9.28 mg L^{-1}, 5.11 mg L^{-1}, 1.18 mg L^{-1}, 51.87 mg L^{-1}, and 1.72 mg L^{-1}, respectively. Peak concentrations were often associated with direct fertiliser input into the stream channel. However, less than 0.5% of the total fertiliser applied was exported from the catchment via stream flow. The short-term nutrient increases occurred either directly after application or during high rainfall events in the immediate post-application period. Overall, elevated nutrient concentrations persisted for anywhere from several days to several months after application and were not considered by the authors to both pose either a eutrophication or human health risk (Neary and Leonard, 1978).

Forest fertiliser use in Australia has evolved from initial applications of trace elements in the 1930s and 1940s to address nutrient deficiencies to the use of fertilisers to improve stand growth. Nitrogen and phosphorus are the two main deficient nutrients in soil, but zinc, potassium, copper and boron can also be lacking (May et al., 2009). Overall, the contribution of forest fertiliser use to nutrient leaching is estimated to be low (0.2% for N, 0.1% for P) perhaps due to the increased focus on improving fertiliser application techniques, diagnostic methods and modelling to optimise fertiliser use (May et al., 2009).

In south-east Australia an 18-year-old *Pinus radiata* stand with a 30 m Eucalyptus riparian buffer was thinned and treated with aerially applied phosphate fertiliser (100 kg P ha^{-1}), followed by nitrogen fertiliser (139 kg N ha^{-1}) two years later (Hopmans and Bren 2007). Median P concentrations increased from 0.002 mg L^{-1} to 0.010 mg L^{-1} in the 6-month post application period before declining to pre-treatment concentrations. Median concentrations of total N and nitrate-N increased from 0.10 mg L^{-1} to 0.15 mg L^{-1} and 0.04 mg L^{-1} to 0.07 mg L^{-1} respectively in the first six months after application. There was also a significant increase in phosphorus in sediment; < 0.9% of total N (over 2.5 years) and 0.7% of total P fertiliser (over 5 years) were exported via the stream (Hopmans and Bren 2007).

Binkley et al. (1999) summarised information from studies of forest fertilisation around the world and evaluated the effects on streamwater

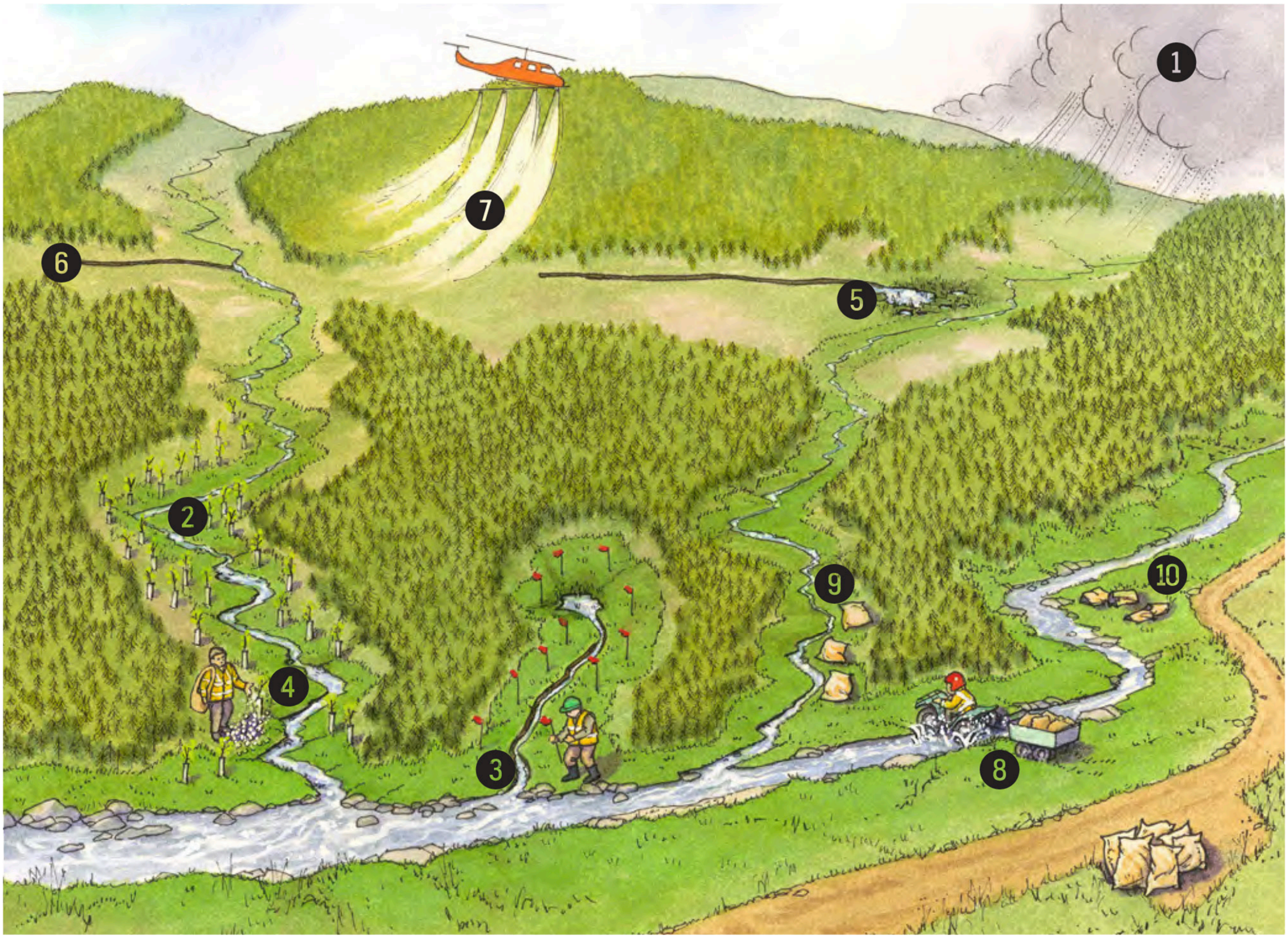

Fig. 3. Best Management Practices to protect water quality from fertilisation. 1. Not applying fertiliser during wet weather (or if heavy rain is forecast within 48 h), if wind conditions are inappropriate, or if the ground is waterlogged, frozen or snow-covered; using the most effective and efficient methods for the site to optimise the amount of fertiliser applied. 2. Not applying organic fertiliser within buffer/RMZ areas. 3. Not applying inorganic fertilisers within defined exclusion zones or buffer areas of any surface water, spring, well or borehole. 4. Restricting the use of inorganic fertiliser within buffer areas to hand applications. 5. Not applying fertiliser when run-off from drains is sufficient to produce visible surface flow across buffer areas. 6. Treating drains that have become sizeable and stable watercourses, and those that flow directly into streams (including road drains), as natural watercourses with their own buffer/RMZ areas. 7. Increasing minimum buffer/RMZ widths for aerial fertiliser applications to land draining to nutrient-sensitive waters, or other identified sensitive areas. 8. Not fording streams with loaded quads or other vehicles when distributing fertiliser bags or other materials around a site. 9. Not storing fertiliser within buffer areas or sensitive areas such as native vegetation, historic sites and neighbouring properties. 10. Not burying or leaving empty fertiliser bags on site; disposing of bags off-site following environmental regulations. *Image reproduced from* Forestry Commission (2019) *© Crown Copyright.*

chemistry. In general, they reported that peak concentrations of nitrate-N in streamwater increased after forest fertilisation, with a few studies reporting concentrations as high as 10 mg N L^{-1} as nitrate, but some of these sites were nitrogen saturated, which explained the higher N losses. For phosphate, peak concentrations > 1 mg L^{-1} were found but annual means were < 0.25 mg L^{-1}. The highest average concentrations of nitrate-N were reported at 4 mg N L^{-1}. Relatively high concentrations were associated with the use of ammonium nitrate rather than urea and with repeated applications although other studies have reported no such cumulative response (Shah et al., 2021). Large peaks in ammonium-N may also occur following fertilisation (up to 15 mg N L^{-1}), but annual averages remain < 0.5 mg N L^{-1} (Binkley et al., 1999).

In their review, Binkley et al. (1999) highlighted the dearth of data on fertilisation in tropical forests. Since then, some data has become available from South East Asia primarily associated with palm plantations. Nitrogen, phosphorus and potassium, applied to oil plantations because of the low soil fertility, are at risk of leaching in tropical climate conditions due to high temperatures, frequent high rainfall events, and high carbonic acid content in the soil (Comte et al., 2012).

Studies in Borneo attributed nitrate increases (Luke et al. 2017) and higher concentrations of phosphorus and potassium (Chellaiah and Yule, 2018) in streams draining palm plantations to fertiliser applications. Nitrogen (ammonium chloride) and phosphorus (muriate of potash) fertilisers were applied to mature oil palm plantations during the monsoon season in East Malaysia to measure leaching rates, and effects on groundwater (Ah et al., 2009). The concentrations of ammonia-N, nitrate-N and K in ground water ranged from 0.23 to 2.7 mg L^{-1}, 0.07 to 0.25 mg L^{-1} and 0.63 to 9.54 mg L^{-1}, respectively. The authors considered that groundwater quality was not affected by these fertiliser treatments, applied at optimal rates for mature oil palms, and concentrations were within water quality guidelines.

2.2.1. *Summary of fertilisation effects*

Forest fertilisation often leads to nutrient runoff particularly in the first few months after application. Although maximum concentrations can exceed water quality standards, annual means are usually well below statutory limits. In most cases, elevated concentrations in streamwater are short-lived and are often associated with rainfall events. Direct application to watercourses poses the biggest risk but this is less common where BMPs are employed (see Section 2.2.2).

Nitrate losses are generally low, particularly in nitrogen-limited areas, however higher concentrations have been recorded in areas that are nitrogen saturated. In general, nitrogen losses appear to be higher from N. American forests compared to European and Australasian areas. The opposite appears to be the case for phosphorus, with higher concentrations reported from Europe than N. America, perhaps due to the lower adsorption capacity of peaty north-west European soils. There is a dearth of information from S. America and Africa although indications are that Eucalyptus plantations in the former make efficient use of fertiliser nutrients and so losses are low.

2.2.2. BMP for forest fertilisation

The use of buffer areas and improvements to aerial targeting systems have reduced the impact of forest fertiliser applications (Binkley et al., 1999; Nisbet, 2001; Nisbet et al., 2002).

Minimising the fertiliser amounts applied can also reduce impacts. In south-east Queensland, a fertiliser application had no discernible effect on total N and P concentrations in stream water and groundwater (Bubb et al. 2002). The authors attributed this result to the low fertiliser application rates (mono-ammonium phosphate applied at a rate of 25 and 50 kg ha^{-1} of N and P respectively), protective buffers along water courses (10 to 30 m wide), and the propensity for P to immobilise in soils. The review by May et al. (2009) also noted that BMPs such as retention of forest litter after harvest and riparian buffers were an effective tool in minimising nutrient run-off and direct fertiliser inputs into water ways.

Fig. 3 highlights some of the BMPs that can help mitigate the effects of fertilisation on water quality.

2.3. Harvesting

Of all the activities that are undertaken in a managed forest, harvesting operations have the greatest potential to adversely impact on water quality. The use of heavy machinery, size of area clearfelled, presence of buffers, topography, soil type and local environmental conditions, particularly meteorological conditions, are some of the major factors that affect the scale of the impact. Other factors include the revegetation rate following harvesting, pre-harvesting soil fertility, and soil buffering capacity (Feller, 2005). A great number of studies have investigated the effects of harvesting on water quality. Here we present some of the literature by region and synthesise the results attempting to cover a wide geographical area, including studies on both clearfelling and thinning (see also Table 2, Appendix A).

Europe

At Plynlimon in Wales, the effects of harvesting on water quality have been monitored in several catchments where the scale of conifer felling varied. The main response to felling was an increase in K and NO_3 concentrations, which declined after a few years following replanting and development of the next generation forest (Neal et al., 2011). Smaller increases in DOC and possibly Al and acidity (lower pH and lower Gran alkalinity) occurred soon after felling in one of the catchments, but the changes were insignificant when compared with annual variability.

Increased K and NO_3-N were found in streams at another Welsh site (Beddgelert) following conventional harvesting but there was no K release from WTH harvested areas, which the authors attributed to the lack of brash (Stevens et al., 1995). Nitrate increases were also reported at Balquihidder in Scotland, where mean nitrate concentrations increased from 20 to 40 μeq L^{-1} (max. of 80 μeq L^{-1}) but declined rapidly after felling and replanting (Harriman and Miller, 1994). At Beddgelert, K increases lasted for 4 years after felling, and NO_3-N for 3 years (Stevens et al., 1995).

Longer durations were reported after clearfelling on peatland at Flanders Moss in Scotland where increased PO_4-P concentrations persisted for 3–5 years after felling ended, with high phosphate concentrations (maximum 1729 μg L^{-1}) partly attributed to decomposition of forest residues including brash (Shah and Nisbet, 2019). The authors also found increased DOC, colour and suspended sediment following conventional stem only harvesting and although NO_3-N concentrations increased after harvesting, they were at a low level.

Clearfelling on deep peat in Ireland led to significant (molybdate reactive) phosphorus increases (9 μg L^{-1} to 265 μg L^{-1}) following partial harvesting of a 100 ha catchment, with a greater increase found when a smaller 1 ha catchment was completely felled (from 13 μg MRP L^{-1} to a peak of 4164 μg MRP L^{-1}) (Cummins and Farrell, 2003a). The results indicate that the proportion of catchment felled is perhaps more important than the area felled. The authors also found increases in alkalinity and concentrations of NH_4-N, NO_3, K, Mg, DOC and organic monomeric aluminium; pH increased at one site only, something that Shah and Nisbet (2019) also found in Scotland. Several reasons were suggested for the pH increase including base cations input from adjacent roads and release of soil organic carbon (DOC) due to a rise in the water-table (Cummins and Farrell, 2003a; Shah and Nisbet, 2019). Clearfelling did not affect concentrations of sulphate, suspended solids or inorganic monomeric aluminium, whilst concentrations of Na, Cl, and Mg, and conductivity were all reduced after felling (Cummins and Farrell, 2003b).

Nutrient and sediment releases were reported at another site in Ireland; during storms, peak values of TSS concentrations increased by up to 50 times the pre-felling concentrations with the magnitude dependant on the percentage of the coupe clearfelled at the time of the storm (Kelly-Quinn et al., 2016). Deep rutting in the clearfelled area was given as a possible reason for the high TSS transport.

In a global analysis (51 catchments including 16 controls) Bathurst and Iroume (2014) found no apparent general relationship between sediment yield impact and the proportion of catchment logged. Their analysis provided quantitative generalisations of the effect of logging on sediment yield. They concluded that for low-moderate and high impacts, the annual specific sediment yield in the logged catchment exceeds that in the control catchment by no more than an order of magnitude. For very high impacts, annual yields may be two orders of magnitude higher. They also found that two thirds of logged catchments deliver their maximum post-logging sediment yield in the first two years after logging.

A recent review of how forest management in Fennoscandia impacted streams found that impacts were from multiple stressors over the entire rotation period and included thinning and ditch maintenance, but final harvest and associated factors such as site scarification and road construction had the largest potential to impact water quality (Kuglerová et al., 2021). Many of the studies of final felling have been on soil solution chemistry in field experiments where other large-scale manipulations such as fertilisation and wood-ash recycling have been the main focus.

Nitrogen losses to surface waters have been found after forest harvest in the boreal zone but the results are variable with some sites reporting no change after logging (Kreutzweiser et al., 2008). Findings from seven field experiments with conventional harvesting showed that site quality impacted the results (Futter et al., 2010). At high productivity sites, the NO_3-N concentrations were higher but the duration of the losses was lower; the authors concluded that overall forest harvesting in Sweden is a minor contributor to N pollution in the Baltic contributing about 3% of the overall Swedish N load to the Baltic (Futter et al., 2010). This conclusion is supported by a Latvian study where no increase of nitrogen concentrations in streamwater was observed in the two years after harvesting, which the authors concluded was most likely related to N attenuation by the forested buffer between the harvested areas and the stream (Libiete et al., 2017).

In another Swedish study clearfelling led to increased concentrations of K, NH_4, NO_3, org-N and tot-N, whilst concentrations of H^+ decreased. At the end of the eight year study period, run-off and chemical composition of the streamwater successively returned to pre-cut conditions (Rosén et al., 1996).

In Finland, long-term water quality monitoring showed that clearfelling increased the export of total N, Total Organic Nitrogen (TON), NO_3-N, PO_4-P, and SS; however, except for PO_4-P and SS, the increases occurred only in the catchment with highest percentage of felling (34%) (Palviainen et al., 2014). This led the authors to conclude that when a small proportion of the catchment is clear-cut and wide buffer zones are left along the streams the excess load of substances into watercourses is minimal, with increases only being significant when the area of clear cutting exceeds 30% of the catchment area (Palviainen et al., 2014).

Studies have linked increased carbon and iron transport or brownification of waters to harvesting and afforestation, partly due to an increase in forest biomass and therefore the soil carbon pool (Škerlep et al., 2019; Finstad et al. 2016; Nieminen et al., 2021). Others have reported that carbon losses and brownification have ceased in southern Sweden (Eklof et al., 2021). Also in southern Sweden, Froberg et al. (2007) found that although fresh litter decomposed rapidly, the leached carbon was retained in the lower organic horizons and that the DOC leaching out of the Oe horizon consisted almost exclusively of carbon from the Oe horizon itself. Similar results were reported by Amiotte-suchet et al. (2007) who found that in coniferous catchments, and to a lesser extent in deciduous catchments, DOC produced during litter decomposition contributed to a small part of DOC exports by stream water. At a deciduous forest in Germany, litter derived carbon was found to be of low importance for DOM formation and carbon loss via soil water (Scheibe and Gleixner, 2014). The studies highlight that there is continuing debate around the origin of DOC to streamwaters, its retention in soils and the effects of land-use including forest operations on carbon transport. Importantly, the effects of afforestation and forest operations on carbon transport may not be seen for decades because the build-up of soil organic carbon stocks and DOC export progress slowly (Škerlep et al., 2019), highlighting the importance of long-term data in understanding the effects of land-use on water quality.

The first large scale Swedish clearcut study in a generation occurred at Balsjö. This provided an opportunity to compare sites with and without buffer zones (Löfgren et al. 2009, Schelker et al., 2016). Clearfelling resulted in increased runoff and increased concentrations of Na, K, Cl, total N, total P, and suspended material, whilst NO_3 leaching only occurred at the site without a buffer, highlighting the value of a buffer for water quality protection (Löfgren et al. 2009). Dissolved organic carbon concentrations also increased in the years after harvesting, especially during spring flood (Schelker et al., 2012). More recent felling at the same site resulted in a fifteenfold increase in NO_3 concentrations in first-order streams but only subtle responses could be detected in third-order streams suggesting that significant dissolved inorganic nitrogen retention occurred between the harvested areas and downstream monitoring sites (Schelker et al., 2016).

One surprise from Balsjö was that a marked effect on mercury concentrations was not found. The increase in flow in the first few years after harvest, however, meant that there was still an increase in mercury export (Sørensen et al., 2009, Eklöf et al., 2014). The concern about mercury stemmed from studies, many from the boreal zone, that suggested forest harvesting increased the concentration and loading of mercury, especially the highly toxic methylmercury (Bishop et al., 2009, 2020; Kronberg et al. 2016; Skyllberg et al. 2009). The effects of harvesting on mercury, however, has proven highly variable (Eklöf et al., 2016) with some studies such as at Balsjö and a Norwegian harvest study (de Wit et al., 2014) finding little effects on concentrations, while others have found large effects (Porvari et al., 2003). Investigating the effects of mercury on fish, Wu et al. (2018) found that forest harvest led to increased mercury levels in fish although there was great variation between the sites monitored.

To increase the yield of biofuel from forestry, WTH and stump harvesting have been considered in the European boreal zone (Börjesson et al., 2017). In Finland, Kaila et al. (2014) studied P transport following harvesting (stem only, WTH and stump harvesting) of Scots pine dominated stands on peatland. They found a variation of responses from no increase to an increase of over 1.5 kg ha^{-1} in the outflow waters. The most significant factor explaining the variation in P loads and concentrations from clear-felled peatland catchments was the post-harvest water level position; the higher the water table, the higher the P export.

In Germany, Georgiev et al. (2021) found that natural forest disturbances and associated salvage logging did not have a harmful effect on the quality of the streamwater within drinking water catchments.

In contrast to most studies, in Turkey, Gökbulak et al. (2008) found that harvesting (11% thinning) led to decreased colour, turbidity, temperature, pH, and electrical conductivity in the treated streamwater, and that sediment concentrations were unaffected.

Effects of harvesting on stream temperature have also been reported. In Wales, bankside felling resulted in 0.7 to 1.2 °C decreases in temperature in January/February, increases of up to 1 °C in May-June, and around 0.5 °C in September-October; the authors concluded that these temperature effects could have significant impacts upon stream ecology (Weatherley and Ormerod, 1990). Stott and Marks (2000) reported an increase in monthly mean and maximum temperatures in streams in following clearfelling in Wales, particularly in the summer, but the effects on the biological status of the stream were unclear.

North America

Shepard (1994) conducted a review of the effects of forest management, including harvesting, on surface water quality in wetland forests of the USA and reported that many of the studies reviewed observed increased concentrations of suspended sediment and nutrients following silvicultural operations when compared with undisturbed controls. Water quality criteria were rarely exceeded by silvicultural operations with water quality parameters returning to undisturbed levels within a period ranging from months to several years. The findings are in line with long-term monitoring in the Southern Appalachians, USA, which showed little effect of clearfelling operations on water quality (Swank et al., 2001).

Conversely, long-term data from an experimental forest in Oregon, USA, indicates that sediment and bed load increased significantly after a 1966 harvest; annual suspended sediment yields returned to pre-treatment levels in the first two decades following treatment, yet bed-load yields remained high throughout the duration of the study (Safeeq et al., 2020). The authors warned against over-attributing effects to historical logging methods, although other studies have confirmed the beneficial effects of improved practices (see 2.3.2 below).

Another long-term study (24 years) in the Pacific Northwest found that nitrate + nitrite and orthophosphate increased in streams after clearfelling and that concentrations were lower in downstream watersheds due to dilution and nutrient assimilation effects (Deval et al., 2021). Any increases in orthophosphate concentrations were attributed the increased streamflow that followed clearcutting. Concentrations also increased in the control watersheds, albeit to a lower extent, which the authors attributed to climate variability or subtle forest succession changes. Overall, the nutrient load was relatively small suggesting that regulations and BMPs were effective in minimising the delivery of particulate-bound pollutants (Deval et al., 2021).

Nitrate increases were also reported by Harr and Fredriksen (1988) in Oregon, USA, who found NO_3-N concentration increases where logging residues were left to decompose naturally. Concentrations increased more than sixfold and commonly exceeded 100 μg L^{-1} during the October-June high-flow season for seven years after logging and were still elevated at the end of the study, 10 years after timber felling. Where logging slash was broadcast burned, NO_3-N concentrations increased roughly fourfold, but rarely exceeded 50 μg L^{-1} and increases had mostly disappeared six years after slash burning. Annual maximum stream temperatures increased by 2 to 3 °C after logging, but increases had largely disappeared within three years (Harr and Fredriksen; 2008).

In the Catskill watershed, New York, McHale et al. (2008) found large increases in NO_3 (>900 μmol L^{-1}) and concurrent releases of inorganic monomeric aluminium (Al_{im}) after clearcutting (80% of the basal area of the catchment); the increased NO_3 could be accounted for by the decreased uptake of the felled trees.

In contrast to the above studies, there was no significant increase in turbidity, TSS, NO_3-N, NH_4-N, PO_4-P, K, and SO_4-S levels after whole commercial conifer harvesting, with partial harvesting and implementation of BMPs given as reasons for the prevention of degradation (Jones et al., 2013). In a review of paired catchment studies, Neary (2016) showed that of 30 paired catchment studies monitoring NO_3-N after partial or complete clearfelling, only one showed an increase (0.3 to 11.9 mg L^{-1}) that exceeded the international water quality standard (10 mg L^{-1}), which was most likely due to the suppression of vegetation regrowth by herbicide treatment.

In the Canadian Boreal zone, Winkler et al. (2009) reported short term increases in DOC and Total P after harvesting that did not impact ecology in receiving lakes. In British Columbia, Scrivener and Brownlee (1989) reported an increase in streambed fines after harvesting but no increase in suspended sediment.

A number of studies on streamwater temperature have been conducted in Oregon, USA. Brown and Krygier (1970) found that annual maximum stream temperatures increased from 13.9 °C to 29.4 °C after clearcutting and Johnson and Jones (2000) reported that maximum stream temperatures increased 7 °C after clear-cutting and burning in one basin and after debris flows and patch-cutting in another. Stream temperatures in both basins gradually returned to pre-harvest levels after 15 years; important factors influencing the temperature increases were removal of riparian vegetation, and conduction between stream water and nearby soils or substrates (Johnson and Jones, 2000). Also in Oregon, Groom et al. (2011) indicated that retaining buffers reduced temperature increases in streams following harvesting. Similarly, Gomi et al. (2006) found in British Columbia that daily maximum temperature in summer increased by up to 2 °C to 8 °C after harvesting in streams where no riparian buffer was retained.

South America

Rodrigues et al. (2019) observed that clear-cutting in Eucalyptus plantations increased sediment concentrations in the streams especially in the post-harvest year; however, the concentration of nutrients was not affected by harvesting. After harvesting eucalyptus plantations in Brazil, exports of suspended solids increased in the year following felling (Câmara and Lima, 1999), as did exports of nitrate, potassium, calcium, magnesium, iron and suspended solids (Vital et al., 1999).

In central Chile, conversion of some areas to forest plantations has resulted in reduced water quality due suspended sediment increases associated with harvesting of *Pinus radiata and Eucalyptus spp*. (Oyarzun and Peña, 1995). Sediment fingerprinting showed that changes in sediment source are closely related to disturbance by clearcutting and the amount of post-cutting rainfall (Schuller et al., 2013). Also in Chile, Fierro et al. (2017) found that increased electrical conductivity and concentrations of nutrients and suspended sediments in streams changed the composition of aquatic invertebrate communities.

Africa

We struggled to find studies on harvesting and water quality from Africa. However, one study comparing sediment and nutrient transport from different land uses concluded that Eucalyptus plantations were effective in controlling runoff, sediment and sediment-associated nutrient losses in Ethiopia (Girmay et al., 2009).

Australasia

In Australasia, harvesting typically involves clearfelling of mature forest stands up to the stream edge where there is no pre-existing buffer, using hauler systems on steeper land and ground-based systems on flatter topography. The trees are extracted to a skid or landing site for processing, although some mechanical processing at the stump occurs on flatter topography.

Of all the forest activities undertaken in managed forests in New Zealand, clear-cut harvesting to the stream edge has the greatest impact on water quality (Baillie and Neary, 2015). The highest risks are associated with clear-cut harvesting on steep erosion-prone land and the associated increases in sediment, particularly from surface erosion and landslides generated during high rainfall events (Basher et al., 2011; Phillips et al., 2018).

One study in New Zealand (Fahey and Marden, 2006) found an eightfold increase in sediment yield following harvesting of a hill-country catchment. In the South Island of New Zealand, suspended sediment was monitored in three catchments with varying intensities of forest harvesting (Basher et al., 2011). Maximum turbidity measured during storm events ranged from 29.8 to 42.0 NTU, maximum suspended sediment concentrations ranged from 1872 to 6142 mg L^{-1} and there was an overall fivefold increase in suspended sediment yields associated with forest harvest activities. Sediment yields typically declined to pre-harvest levels within 2 to 6 years as vegetation re-established (Fahey and Marden 2006; Basher et al., 2011; Phillips et al., 2018).

Campbell and Doeg's (1989) review on the effects of harvesting on water quality in Australia also highlighted the risks associated with post-harvest erosion and run-off. Given the high clay content in Australian soils, the risk of elevated fine sediment and turbidity in watercourses from harvesting activities is high. The results of this review typically show increased turbidity and suspended sediment following harvest operations, with the timing and intensity of rainfall events following harvest having a strong influence on the amount of sediment entering the watercourse. In contrast, Hancock et al. (2017) found no difference in sediment loads between harvested and control catchments in south-east Australia, concluding that tree harvesting and subsequent BMPs employed do not produce detrimental effects in the medium to long term.

Along with sediment, a pulse of nutrients, particularly nitrates, often occurs after harvest as the removal of forest cover increases nutrient leaching into waterways (Campbell and Doeg, 1989; Baillie and Neary 2015). Both Graynoth (1979) and Thompson et al. (2009) recorded increases in average nitrate concentrations (range 0.12 to 0.36 g m^{-3}) following clear-cut harvesting in the South Island of New Zealand.

However, not all harvest operations generate increases in stream nutrients. In Queensland, Australia, 80 ha of 36-year-old trees were clearfelled, and 250 ha of 24-year-old trees were thinned using mechanical harvesters (Bubb et al. 2002). The streamflow-weighted concentrations of Total N, Total P and SS at two stream monitoring stations ranged from 0.36 to 2.44, $<$0.01 to 0.28 and $<$ 10 to 264 mg L^{-1}, respectively. The similarity in the Total N, Total P and SS concentrations at both stations throughout the study indicated that these harvest treatments had minimal impact on water quality. Overall, nutrient increases following forest harvest were typically short-lived ($<$2 years) as vegetation re-established, or indiscernible from pre-harvest or control site concentrations (Campbell and Doeg, 1989; Baillie and Neary 2015).

Removal of shade and increased light levels during clear-cut harvesting, frequently results in increased stream temperatures, particularly in smaller-sized streams. In New Zealand, maximum temperatures up to 25 to 30 °C have been recorded following forest harvest to the stream edge, with diurnal ranges up to 12 °C (Baillie et al., 2005; Quinn and Wright-Stow 2008). Harvesting impacts on water temperature were generally greater in smaller streams that lack the flows and thermal insulation of larger streams and rivers. However, their capacity for recovery was greater, often within a few years of harvest, whereas streams in larger catchments or where there is progressive harvesting over a period time can take up to 8 to 10 years or more to recover (Quinn and Wright-Stow 2008; Baillie and Neary 2015).

Asia

Similar to other regions around the globe, sediment is the dominant water quality issue following forest harvest in Asia. For example, in the hilly terrain in Sabah, Malaysian Borneo, a trial assessed the effects of varying forest disturbance on sediment (Nainar et al., 2017). In general, there was an increase in mean discharge-weighted suspended sediment concentrations (SSC) and annual sediment yields with oil palm having a far greater impact on sediment than the other forest types (primary forests and jungle reserve) for both annual and storm event data. The authors attributed this to the bench-terraced slopes, higher density of roads and tracks, gullying and a lack of riparian buffers.

Douglas et al. (1992) assessed the different stages of logging on suspended sediment in a 54 ha forest catchment in Sabah over a 27-month period. Harvesting within the vicinity of the road increased monthly SS yield fivefold compared to the control site, with a peak sediment concentration of 12,947 mg L^{-1} measured in the first storm event. Monthly SS yields increased eighteenfold after logging the rest of the catchment, with SS concentrations of over 1000 mg L^{-1} in most storm events. Sediment level were starting to recover one year after harvesting was completed. The study highlights the importance of capturing sediment data during storm events.

Asian studies covering a wider range of water quality parameters indicate varying responses to harvesting activities. In Sarawak, Malaysia, the majority of forests have been subject to logging (Ling et al. 2016). The largest effects of logging and canopy removal were seen on temperature variation and sedimentation. Mean water temperatures ranged from 24.7 to 28.8 °C with temperatures in the unlogged sites significantly lower with most, but not all the actively logged sites. Overall TSS concentrations were low and TSS concentrations were lowest at the control sites (≈ 2 mg L^{-1}) compared with actively logged sites (10 to 16 mg L^{-1}). Mean turbidity was low in all streams (range 1.5 to 7.7 NTU) with higher turbidity in the active logging sites compared with the control sites. The results for Total N, total ammoniacal-N, nitrate-N, Total P and soluble reactive phosphorus were variable with no clear differentiation between the control and active logging sites (Ling et al., 2016).

A study in Malaysian Borneo compared water quality in old-growth dipterocarp forests, selectively logged forests, and oil palm plantations with and without riparian buffers (Luke et al. 2017). These categories provided a decreasing gradient of 'forest quality' for analysis. Water temperature showed a significant increase along this gradient (mean 24.99 °C, 25.02 °C, 26.86 °C, 28.22 °C respectively). Concentrations of Nitrate-N and reactive-P were well below pollution threshold levels. There were no significant trends identified for dissolved oxygen, pH, conductivity. However, water quality had not fully recovered 10 to 15 years after select harvesting.

Similar to Douglas et al. (1992), Oda et al. (2011) showed the importance of sampling storm flows along with base flow to estimate the effects of clear-cut harvesting on stream water quality. Results from the base flow sampling showed that increased nitrate concentrations peaked one year after harvest (28 μeq L^{-1}) and decreased gradually over the next 6 years. In the storm event sampling, nitrate concentrations increased with discharge, peaking at 700 μeq L^{-1}, one year after harvest. K, Na, and Ca also increased with discharge, showing that clear-cutting led to the release of cations during high flows. Baseflow data alone markedly underestimated the export of solutes from these catchments.

Tokuchi and Fukushima (2009) used chrono sequencing to assess the long-term effects of clear-cut harvesting on water quality in central Japan. The study included 40 catchments with similar characteristics, covering 1000 ha and the plantation stands ranged in age from 0 to 87 years. There was a strong positive relationship between nitrate concentrations in stream water and stand age in the first few years after harvest. Across the age classes, nitrate showed a clear relationship between concentration and stand age, indicating that nitrate concentrations were mainly regulated by vegetation regrowth and that clear-cut harvesting influenced nitrate concentrations for several decades. In contrast, SO_4, Ca, Mg, Cl, and Na concentrations appeared to be controlled by catchment characteristics such as geology and topography. The exception was K where rainfall was the likely controlling factor for the concentrations of K found in stream water.

A number of studies in Japan have assessed the effects of thinning on water quality. Nam et al. (2016) found that thinning (50% strip thinning) led to increased suspended sediment yields but the results were confounded by sediment increases during storms. In Tochigi, Japan, strip thinning significantly increased dissolved total phosphorus, total phosphorus and dissolved organic carbon (0.01, 0.04, 0.53 mg L^{-1} respectively) in stream water during the thinning and dissolved total nitrogen and total nitrogen (0.34 and 0.46 mg L^{-1}, respectively) after the thinning, relative to the un-thinned basins (Fukushima et al., 2015). The proportional increase in particulate-P and to a lesser extent particulate-N, along with increases in DOC, suggested increased surface and subsurface flow from soil disturbance from the strip thinning operations. The increase in dissolved nutrients particularly dissolved-N was attributed to increased leaching post-thinning.

In northern Kyushu, western Japan, Chiwa et al. (2020) assessed the effects of thinning a nitrogen saturated plantation (43% of basal area) on water quality. Water samples taken during baseflows and stormflows before and after thinning showed minimal changes in dissolved organic-N (35.8 kg N ha^{-1} before and 36.5 kg N ha^{-1} after thinning). Exports of N during storm flow were slightly higher after thinning (10.5 kg N ha^{-1} before and 12.2 kg N ha^{-1} after thinning) as the proportion of N exported increased with the increased water yields after thinning.

A study in Japan assessing the effects of strip thinning on water temperature found significant increases in mean (11.5 °C before thinning, 14.8 °C during and 12.2 °C after strip thinning) and maximum (22.3 °C before thinning, 22.4 °C during and 26.2 °C after strip thinning) water temperatures which were positively correlated with solar radiation and negatively related with discharge (Oanh et al. 2021).

2.3.1. Summary of harvesting effects

Following harvesting, increased sediment delivery, nutrient transport and DOC increases are widely reported. Of the nutrients, the highest and most persistent increases are found for P with concentrations sometimes exceeding European standards (Shah and Nisbet, 2019) and remaining elevated for up to 3 to 5 years (Rodgers et al., 2010; Shah and Nisbet, 2019), but even for as long as 14 years (Palviainen et al., 2014). K leaching is generally lower than P and for shorter durations. In Europe, nitrate increases are usually short-lived and low level, not exceeding water quality standards, but in North America and Australasia (New Zealand) increases are more pronounced. The difference is partly due to the differing environmental conditions, including the soil type and perhaps the deposition chemistry and recovery from historical acidification.

For pH, both increases and decreases are reported in Europe and may be related to the base cation concentrations in stream waters following harvesting. Increased trace metals have been reported after harvesting, but concentrations are low and short-lived. High concentrations of sediment delivery and suspended solids are commonly reported after harvesting, but usually for short-durations and often associated with extreme meteorological conditions, especially heavy precipitation or rainfall after a dry period. For DOC, the main increases are usually found in the first years after harvest and usually only at the local scale (Futter et al., 2016).

Streamwater temperature increases are commonly reported after harvesting with short-term and long-term (15 years) impacts reported.

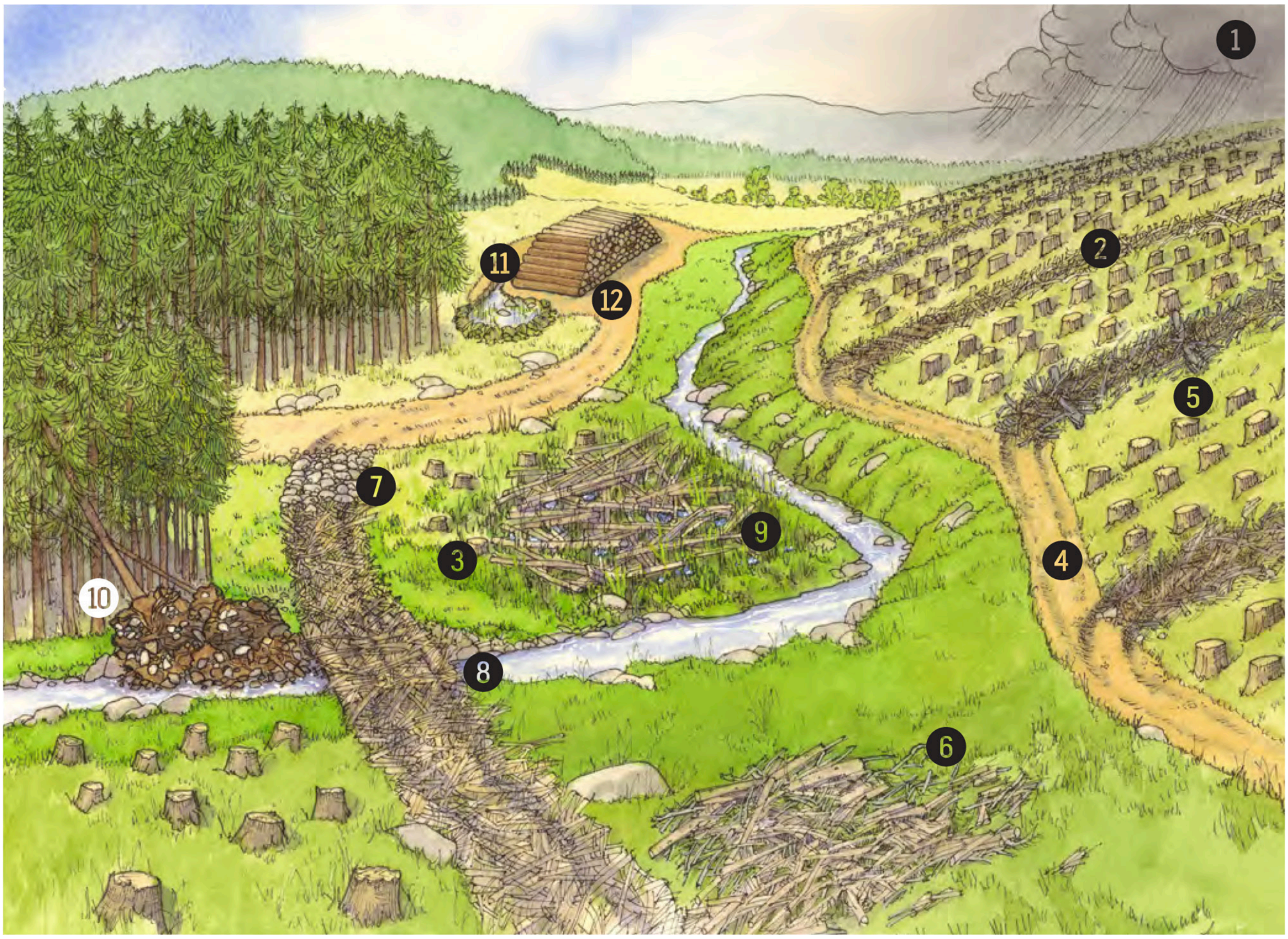

Fig. 4. Best Management Practices to protect water quality from harvesting operations. 1. Monitoring weather forecasts daily and amending work plans accordingly. Suspending operations during heavy rainfall or when ground is saturated. 2. Avoiding long, straight extraction routes and ensuring brash/slash mats are maintained. Limiting machine operations on slopes > 20% (11.3°). Lowering the ground pressure of machines. 3. Avoiding using skidders on soft ground; locating skid trails away from streams and closing them quickly. 4. Keeping extraction routes outside buffer/RMZ areas and valley bottoms wherever possible. 5. Avoiding skidding on long steep slopes, aiming for < 20% (11.3°); carrying out skidding at an angle to the slope rather than straight up and down a hill. Using log steps where rutting occurs to split run-off and diverting it to unbroken ground. 6. Having a brash/slash management plan in place to avoid build-up of brash/slash in high-risk areas where it could potentially mobilise (for example, steep unstable topography in the vicinity of waterways and around landings). Ensuring run-off from brash/slash does not drain directly into watercourses. 7. Using stone ramps to protect main access routes. 8. Protecting stream crossings from damage to stream banks and beds; constructing bridge, culvert or pole crossing at elevations higher than the road approach. 9. Considering felling crops but not extracting timber where this would cause major damage to very soft ground (poor growth areas on peat, for example). Undertaking phased felling to reduce soil exposure and disturbance, and changes to drainage water chemistry. 10. Avoiding exposing conifer crops on the bank of a watercourse opposite the felling site, where these are vulnerable to windblow. Where practical, trying to replace any upturned root plates to restore banksides. 11. Ensuring run-off from roadside timber stacks and loading areas does not drain directly into watercourses; disconnecting road drains. Avoiding landings within 50 m of a watercourse. 12. Suspending operations if heavy rainfall leads to a build-up of mud on timber stacking and loading areas, especially where there is a risk of run-off reaching local watercourses. *Image reproduced from* Forestry Commission (2019) *© Crown Copyright.*

The retention of riparian buffers and speed with which riparian vegetation recovers are major factors affecting the spatial and temporal impact of harvesting on streamwater temperature and chemistry.

The main impacts of harvesting on water quality are often short-lived with recovery in the range of months to within 4 years; however, several studies reviewed in this section have highlighted that long-term impacts do occur for up to a decade or more. The results emphasise that in some cases long-term studies are required to understand the full impact of forest management on water quality.

2.3.2. *BMP for forest harvesting*

Fig. 4 highlights some of the BMP measures that can help to protect water quality from harvesting operations. A wide range of measures are now available to forest managers with some of the most effective ones highlighted below.

Buffers/Riparian Protection Zones

The retention of riparian buffers can be effective at mitigating many of the adverse effects of clear-cut harvesting on water quality. A comparison of the effects of felling on suspended sediment pre and post implementation of BMPs recommended in the UK Forests and Water guidelines found that suspended sediment loads decreased, primarily due to the establishment of a 50 m buffer zone (Stott et al., 2020). A large scale Swedish clearcut study at Balsjö compared the effect of a riparian buffer zone to a catchment without a buffer zone; the value of the buffer zone was apparent because NO_3 leached at the site without the buffer but not at the one with the buffer (Löfgren et al., 2009, Schelker et al., 2016).

In the Northwest California, Rice et al. (2004) documented the effectiveness of forest practice guidance finding that suspended sediment loads increased almost threefold from selective logging and road construction prior to implementation of the 1973 Forest Practice Act, and that smaller but statistically significant increases in sediment were associated with clearcutting and roads under forest practice rules in effect since 1990.

In Oregon, USA, Rachels et al. (2020) found that after a 2016 partial harvest BMPs, especially riparian buffers, were effective at protecting streams from sediment delivery. Hatten et al., (2018) conducted a more direct comparison of contemporary and legacy harvests on research watersheds, also in Oregon USA, and found that modern silviculture following BMPs did not result in an increase in suspended sediment.

McBroom et al. (2008a, 2008b) revisited research watersheds in Texas, USA, that were clearfelled in 1981 without BMPs (Blackburn et al., 1986). After modern harvest with BMPs, nutrient losses were generally minor and intensive silvicultural practices with BMPs did not significantly impair surface water quality with N and P (McBroom et al., 2008a). Sediment loss was only significant on three out of the six small clearfelled watersheds, and sediment loss from the watershed with the highest loss rate was one-fifth the loss of the 1981 harvest (McBroom et al., 2008b). While there were other factors, reduced areas of bare soil and improved riparian buffers, especially on intermittent streams, offered enhanced protection.

In Chile, a riparian buffer with native vegetation in pine and eucalyptus plantations proved effective in reducing the export of nutrients; for sediment, results were comparable to reference watersheds (100% native forest) where streamside buffer widths were $\geq$ 36 m, and for nitrogen (Total N and DIN) where buffer widths were 17–22 m (Little et al., 2015). In Brazil, Cassiano et al. (2022) suggested that effectiveness of riparian buffer for sediment retention depends on the ratio of harvested forest to riparian forest proportion (H/F) at the catchment scale.

In Victoria, Australia, a 46.4 ha catchment was cleared of native Eucalyptus forest for conversion into *Pinus radiata* (Hopmans et al. 1987). A 30 m buffer was retained along the stream edge. Water temperature, colour, suspended solids and pH showed no significant change following harvest, but there was a minor increase in turbidity and conductivity. Overall, effects on water quality were low and attributed to the 30 m buffer and high infiltration capacity within the catchment.

In a native forest in New South Wales, Walsh et al. (2020) assessed the effects of catchment harvesting, using BMPs and either select harvesting or no harvesting, in 10 m streamside buffers on turbidity and SS. Harvesting resulted in significant increases in SS yield compared to an unharvested control site, but there were no significant impacts on turbidity. Where buffers were retained, SS concentrations and hydrological flows largely recovered with 18 months, whereas the select harvested buffers had yet to recover.

In New Zealand, the retention of streamside buffers was effective at mitigating water temperature increases and nutrient inputs but had limited ability to filter out sediment, particularly point sources of sediment (Graynoth, 1979; Thompson et al., 2009).

Luke et al. (2017) found that the retention of buffers had variable effects in mitigating the impacts of forest harvest on water quality in Malaysian Borneo. Although water quality was higher in palm plantation sites with buffers, they did not fully protect the stream from management activities.

In Selangor, Southeast Asia, water quality was measured where two catchments were clearfelled for conversion to plantation forest (Marryanna et al. 2007). There were no significant differences in colour due to harvesting. Turbidity and conductivity increased during and after harvesting with the impact greater at the non-buffered site compared to the buffered site.

In contrast to the above, Nieminen et al. (2020) found that where peatland catchments were restored for use as wetland buffer areas as a part of best management practices, restoration induced considerable increases in nutrient, carbon, and heavy metal exports.

Low impact techniques

Low impact logging techniques can effectively protect water quality. In Japan, Hotta et al. (2007) assessed the performance of low-impact logging (skyline logging, strategically placed slash/brash cover to minimise soil disturbance) in steep hill country in reducing suspended sediment (SS) yields commonly associated with harvest. No increase in annual SS yields was detected although water yield did increase.

Low impact management techniques also helped protect water quality after felling on sensitive peaty soils in Scotland; measures included the use of brash and undersized logs to strengthen forwarder tracks and their subsequent removal for use as biomass for fuel, thereby reducing the potential for PO_4-P and DOC leaching (Shah and Nisbet, 2019).

Modifications to machinery and practice can help reduce site disturbance. Labelle et al. (2022) indicated that physical disturbance from forest machinery can be mitigated by restricting machine operating trails to $<$ 20% and by reducing ground pressure of machines through, for example, high flotation tires and use of steel flexibles tracks.

Phased felling

Phased (also staggered or partial) felling involves harvesting an area over several years rather than a short period of time or reducing the catchment area felled. This can be amongst the most effective ways to protect the water environment from felling activities.

At a site in Wales where felling was phased over many years the water quality response was almost unnoticeable (Neal et al., 2004a), an effect that was also reported in Brazil where sediment yield with staggered felling (21% of the catchment area) was similar to the yield prior to harvesting (Valente et al., 2021). Phased felling reduced and even prevented water quality impacts following clearfelling on peatland in Scotland, with PO_4-P increases lower than with conventional harvesting, and very little if any increase in DOC, NO_3-N or suspended solids (Shah and Nisbet, 2019). In the Catskill mountains of New York, no increases in NO_3 or Al_{im} (inorganic monomeric aluminium) were seen in the areas that underwent selective harvesting ($<$10% of the basal area of the catchment harvested), and there was little effect on brook trout survival (McHale et al., 2008).

Clearly, many studies have documented the value of BMPs for water quality protection, and management guidance has played a significant role in implementation. In the UK, for example, the Forests and Water Guidelines and associated guidance has led to significant reductions in water quality impacts (Carling et al., 2001), as have Best Management Practices in the U.S. (Cristan et al., 2016). The issue of BMP compliance remains, however, and issues such as climate change and water provision require the development of more refined management practices that anticipate future impacts upon water quality and ecosystems (Sun and Vose, 2016).

2.4. Conclusions

Across the wide geographical area and varied environmental conditions covered in this paper, sediment delivery was the most frequently found and significant water quality impact of forest management, primarily resulting from cultivation operations, drainage and harvesting. Annual mean concentrations were usually below ecologically damaging levels but peaks, particularly after storms, were at levels that could adversely impact upon aquatic life.

Nutrient losses can be significant but, in comparison to sediment, these appear to be more significantly affected by local conditions including meteorological conditions, soil type, and the management techniques employed such as the amount of forest materials left on-site. In general, forestry contributes relatively little in the way of nutrient loads to the environment particularly when compared to the contribution of other land uses such as agriculture; moreover, impacts are usually short-lived although elevated nutrient concentrations for 5 years or more have been reported.

Organic carbon losses are mainly reported after harvesting operations and have been related to soil disturbance and breakdown of forest materials, although a rise in the water table due to the reduced evapotranspiration following tree felling, and increased temperature after the removal of tree shading have also been suggested as reasons for elevated DOC concentrations. In general, the greatest increases in concentrations and catchment exports are seen in the year after felling although in some studies concentrations have been found to persist for 3 to 4 years or more. Forest management, particularly drainage, is also one of a number of factors influencing long term changes in water colour and organic matter content.

Metal and base cation releases are reported after cultivation, drainage and harvesting but increases are usually short-lived and most likely related to soil disturbance; changes in pH are rare, particularly in the medium to long-term. Temperature effects are significant, with both stream and surface temperatures shown to increase after forest removal; studies indicate that riparian woodland plays an important role in regulating streamwater temperatures for aquatic life.

Spatial and temporal scale are important considerations. Changes in water quality at the local scale are often not seen at the catchment scale highlighting the importance of monitoring at an appropriate spatial resolution. Significant changes in local water quality following forest operations are often reduced or undetectable in downstream sampling points meaning that there will be little in the way of environmental impacts (Deval et al., 2021; Futter et al., 2016; Neal et al., 2004b; Schelker et al., 2016). Monitoring at the appropriate temporal scale is required to fully assess the short and long-term impacts of forest operations; for example, in some areas drainage may affect water quality decades after the operations were carried out (Nieminen et al. 2018b; Finér et al., 2021).

Despite the early forest hydrology studies based in Africa (Blackie and Robinson, 2007; Kruger and Bennett, 2013), there is a dearth of literature on forest management and water quality reported from the region. This is partly due to there being fewer hydrological studies conducted, but there are other factors including an absence of commercial forestry in some countries, a lower level of research output from Africa (King, 2004), and the absence of research publications and journals on citation index listings (Tijssen, 2007). Similarly, there are few studies in South Asia and South America, again, reflecting not only fewer research studies on this topic in these regions but perhaps also inequalities in academic knowledge production, and the lack of accessibility of research in the global south due to mechanisms associated with publication (Collyer et al., 2016). There was also little data from Russia and China, despite their large forest areas; it is possible that the literature exists but is not available in English.

2.5. Looking forward

Opportunities exist to improve the experimental design of water quality experiments to account for the heterogeneity found in forested environments (Akroume et al. 2016). Many studies report results of weekly, fortnightly, or monthly water sampling regimes. These provide a good picture of long-term trends and seasonal averages but can miss short-term event related concentrations and fluxes (Gao et al., 2020). Combining traditional laboratory analysis with high-frequency monitoring sensors provides opportunities to capture temporal changes in water quality that less frequent sampling misses (Neal et al., 2012; Khamis et al., 2021), thereby providing a better understanding of water chemistry dynamics after forest operations. Moreover, fingerprinting techniques (Collins et al., 2017; Rachels et al., 2020) and stable and radioactive isotope analysis (Gibbs, 2008; Schuller et al., 2013) have developed in recent years and provide opportunities to improve land management practice by better understanding the sources and pathways of pollutants.

Improvements in a range of technologies including chemical tracers, drone observations, high resolution mapping, satellite data and advanced remote sensing may help improve our understanding of forest management impacts on hydrology in future, and facilitate more targeted and precise forest management regimes that will benefit water quality (Rubilar et al., 2018; Sims et al., 2013; Sun and Vose, 2016).

One of the key risks to building upon advances in forest management practice is the cessation or reduction of long-term monitoring. Long-term studies are essential if we wish to understand the effects of forest management on water quality at a timescale appropriate for forestry and strategic land management (Burt et al., 2008; Lovett et al., 2007). Moreover, long-term monitoring provides insights into non-forestry related effects on water quality helping to separate out the effects of background changes in climate from land management, for example (Deval et al., 2021; Shah et al., 2021).

Globally, plantation forest cover continues to increase (FAO and UNEP, 2020) and forest managers face increasing public scrutiny on the sustainability of forest operations. Due to the competition for land many forests are being planted in areas difficult to reach with conventional machines and foresters will need to consider the latest harvesting technology, methods and BMPs to mitigate impacts on water quality (Labelle et al., 2022; McEwan et al., 2020).

Finally, the effects of forests and the climate on water quality are inextricably linked, and climatic extremes such as drought and storms may increase nutrient and carbon transport and impact upon freshwater ecosystems (Whitehead et al., 2009). Climate change mitigation strategies such as the expansion of short rotation energy forests or carbon forests with longer rotations and higher stocking rates bring new challenges to the water environment, particularly where afforestation rates are accelerated due to financial incentives (Pawson et al., 2013). These and future challenges can only be addressed by balanced forest management strategies that take full consideration of forest-water-climate interactions and potential impacts.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A

Table 2
The effects of harvesting on water quality reported by selected studies from across the globe[1].

Study	Location (Site name)	Soils	Felling extent	Effects on water quality	Selected data post-harvesting	Comments
Europe						
Cummins and Farrell (2003a)	Ireland (Cloosh)	Blanket peat	33–100% of catchment	Increased PO_4	Reactive phosphorus: Max. 4.16 mg L^{-1} Median > 0.070 mg L^{-1}	Barely noticeable response where felling was phased.
Cummins and Farrell (2003b)	Ireland (Cloosh)	Blanket peat	33–100% of catchment	Increased alkalinity and pH, and concentrations of NH_4-N, NO_3, K, Mg, DOC and organic monomeric aluminium; no effect on sulphate, suspended solids or inorganic monomeric aluminium concentrations.	NO_3-N: Max 0.6 mg L^{-1}	Response related to proportion of catchment felled. Concentrations of Na, Cl, and Mg, and conductivity were all reduced after felling
Gökbulak et al. (2008)	Turkey (Belgrad)	Vertic Xerochrept (Inceptisol/ Brown earth)	11%	Decreased colour, turbidity, temperature, pH, and electrical conductivity; sediment unchanged	Suspended sediment: Max 197.25 mg L^{-1}	Low impacts partly due to large water holding capacity delaying runoff and retarding surface flows to streams; also, low stream water discharge for most of the year except during the intensive rainfall.
Harriman and Miller (1994)	Scotland, UK (Balquihidder)	Peat, gleys, podzols, brown earths	65% of catchment	Increased NO^3; reduced Cl and SO_4	NO_3: Max 1.12 mg L^{-1}	Cl and SO_4 decline most likely due to reduced forest scavenging. Sediment increases related to climate.
Libiete et al. (2017)	Latvia	Folic Umbrisols, Rheic Histosols Albic Arenosols		No significant increase of the dissolved N in streams	NH_3-N: Max 0.27 mg L^{-1} NH_4-N: Max 0.31 mg L^{-1}	N attenuation most likely due to the forested buffer between the clearcut and stream.
Lofgren et al. (2009)	Sweden (Balsjö)	Orthic podsol, Histosols	30% and 73% of catchment	Increased concentrations of Na, K, Cl, total N, total P, and suspended material.	NH_4-N and NO_3-N export: increased from 0.36 to 0.86 kg N ha^{-1} y^{-1} after harvest	NO_3-N leaching increased only from the catchment without a forest buffer. After 8 years streamwater chemistry returned to pre-cut conditions.
Neal et al. (2011)	Wales, UK (Plynlimon)	Blanket peat, gleys, podzols	50–100% of catchment	Increased NO_3 and K, declining after a few years; very little if any increase in DOC and acidity.	K: average 0.24 mg L^{-1} (storm 0.37 mg L^{-1}) NO3: average 2.33 mg L^{-1} (storm 2.71 mg L^{-1})	Barely noticeable response where felling was phased.
Palviainen et al. (2014)	Finland	Iron podzols, peaty podzols, shallow fibric histosols	8–34%	Clear-cutting increased export of total N, TON, NO_3-N, PO_4-P, and SS.	NO_3-N means: 6.8 μg L^{-1} to 42.2 μg L^{-1} NH_4-N means 3.16 μg L^{-1} to 16.3 μg L^{-1} PO_4-P means:1.2 μg L^{-1} to 4.2 μg L^{-1} TOC means: 6.2 mg L^{-1} to 27.3 mg L^{-1} SS means: 0.06 mg L^{-1} to 0.26 mg L^{-1}	Except for PO_4-P and SS, increases were only seen in the catchment with 34% felling. Increases lasted 11 to 14 years.
Rosén et al. (1996)	Sweden	Podzols	50–90% of catchment	Increased concentrations of K, NH_4, NO_3, org-N and tot-N; decreased concentrations of H^+.	Average annual flux: NO_3-N: 0.05 to 0.97 kg ha^{-1} yr^{-1} NH_4-N: 0.12 to 0.52 kg ha^{-1} yr^{-1}	Chemistry of the streamwater returned to pre-cut conditions after 8 years.
Schelker et al. (2016)	Sweden (Balsjö)	Orthic podsol, Histosols	3 to 56%	Increased NO_3, NH_4 and DON	NO_3: Average 261.0 μg N L^{-1}	Increased concentrations in first-order streams but little response in third-order streams.
Shah and Nisbet (2019)	Scotland, UK (Flanders Moss)	Blanket peat	15–100% of study catchment	Increased PO_4-P, DOC, colour and suspended sediment; pH increased at one site. NO_3-N increased slightly	PO_4-P: Max 1.73 mg L^{-1} NO_3-N: < 0.5 mg L^{-1} SS: Max 1085 mg L^{-1}	Reduced impact with phased felling and low impact harvesting techniques; PO_4-P response returned to baseline levels 3–5 years after felling ended; main DOC increase in first year after felling.
Stevens et al. (1995)	Wales, UK (Beddgelert)	Podzols	28 and 62% of catchment	Increased K, NO_3-N	Peak K flux: 29 kg ha^{-1} yr^{-1} NO_3-N: Max 1.8 mg L^{-1}	K concentrations elevated for 4 years from felling. No release of K or P with WTH, attributed to lack of brash.
N America						
Hatten et al. (2018)	Oregon, USA (Alsea)	Loams and gravelly loams on the hillslopes and valley bottoms and clay loams on the ridges.	25% and 86%	No evidence that contemporary harvesting techniques affected suspended sediment concentrations or yields	Suspended sediment yields up to 313 and 102 Mg km^2 yr^{-1} in unmanaged and managed controls, respectively; 127 Mg km^2 yr^{-1} in the treated.	Contemporary harvest with BMPs did not increase sediment above historical levels, unlike legacy (1966) harvests that caused significant increases.

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Table 2 (*continued*)

Study	Location (Site name)	Soils	Felling extent	Effects on water quality	Selected data post-harvesting	Comments
Macdonald et al. (2003)	British Columbia, Canada (Baptiste Creek)	Basal till, silty sand	Approx. 55%	Sediment delivery increased, producing short term and infrequent TSS increases approaching regulatory limits (<24 h, up to 100 mg L^{-1})	Suspended sediment: Peak concentrations > 80 mg L^{-1}	Authors concluded machine free riparian zones would protect streams from bank disturbance.
McBroom et al. (2008a, 2008b)	Texas, USA (Alto)	Clayey, mixed, thermic Typic hapludults with a fine-textured, sandy loam A-horizon and an argillic B-horizon.	Clearfelling with riparian buffers maintained.	Nutrient export did not increase after harvest. Sediment export elevated on an intensive management watershed: increase of 540 kg ha^{-1} but significantly less than the legacy harvest.		Authors noted the benefit of contemporary BMPs in stabilizing riparian areas and preventing direct fertiliser application to streams.
Rachels et al. (2020)	Oregon, USA	Silty clay loam	Relatively small harvest areas according to BMP restrictions.	Mean suspended sediment concentrations were higher in the reference than treated, and the highest storm concentrations also were found in the reference stream.		Study was able to document sediment sources as hillslope, road, and instream.
Scrivener and Brownlee (1989)	British Columbia, Canada (Carnation Creek)		100% of catchment with no riparian buffers	Increased fines, no increased suspended sediment	Fines increased by 4.6 and 5.7%	Increase in fines decreased salmon survival significantly.
Swank et al. (2001)	North Carolina (Coweeta)	Typic hapludult, Dystrochrept, Halumbrept		Small increases in nutrient losses following clearcutting and logging. Responses largest in third year after treatment. Cumulative increase of 200 tonnes in sediment yield in 3-year period after logging.	NO_3: Max 15 μeq L^{-1}	Minimal impact of the management on ecosystem health (long-term benthic studies)
S America						
Câmara and Lima (1999)	Itatinga, São Paulo, Brazil (Tinga reach)	Typic and Rhodic Hapludox	92% of catchment	Increased turbidity, suspended sediments, cations Fe and K. No changes in pH and electrical conductivity	SS: Max 15.9 mg L^{-1} Mg: Max 0.8 mg L^{-1} Fe: Max 1.4 mg L^{-1}	Most effects concentrated in the 4 rainy months after harvesting.
Cassiano et al. (2022)	Itatinga, São Paulo, Brazil 3 catchments	Typic and Rhodic Hapludox	60% to 92%	Increased suspended sediments according to harvest proportion	SS baseflow Max: 6.1 mg L^{-1} SS stormflow Max: 484.1 mg L^{-1}	Authors suggest the ratio harvest/riparian forest area proportions (H/F) as a predictor of suspended sediments at catchment scale.
Mendes et al. (2021)	Parominas, Pará, Brazil (Capim riverbasin)	Ferralsols	Selective logging	Increased water temperature and reduce oxygen at conventional logging catchments compared to catchments with pristine forest or reduced impact logging	Temp Max: 26.3 °C Oxygen Min: 4.9 mg L^{-1}	Stream conditions after reduced impact logging similar to pristine forest, while conventional logging caused impacts on water quality.
Oyarzun and Peña (1995)	Angol, Chile (Picoiquén watershed)	Haplumbrept	1 ha plot treatments	Increased suspended sediments at clear-cutting plots.	SS: Max: 561 mg L^{-1}	Removal of residues and burning practices led to higher concentrations of suspended sediments than control (undisturbed) and logging with residues left on-site.
Rodrigues et al. (2019)	São Paulo, Brazil, 4 catchments	Entisols QuartzipsammentsInceptisols	59% to 91.5%	Increased suspended sediments; different responses for Ca, Mg, NO_3, K	SS: Max 195 mg L^{-1} Ca: Max 8.5 mg L^{-1} Mg: Max 2.2 mg L^{-1} NO_3: 4.9 mg L^{-1} K: Max 6.1 mg L^{-1}	Effects on water quality aggravated or attenuated by natural characteristics such as soil and landscape planning. TSS, and cations and anions direct related to road density, proportion of conservation areas and inversely related to harvested proportion (catchment scale)
Vital et al. (1999)	Santa Branca, São Paulo, Brazil (Bela Vista III)	Hapludulf type	96.7% of catchment	Increased suspended sediments, electrical conductivity, and nutrient export NO_3, Ca, and Fe. No changes in pH, alkalinity, turbidity, and nutrients export K, Mg, Na	SS: Max 48 mg L^{-1} EC: Max 137pt Co NO_3: Avg 1.5 kg ha^{-1} $year^{-1}$ Ca: Avg 5.1 kg ha^{-1} $year^{-1}$ Fe: Avg 5.7 kg ha^{-1} $year^{-1}$	Comparison of effects one year after harvesting to previous 7 years.

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Table 2 (*continued*)

Study	Location (Site name)	Soils	Felling extent	Effects on water quality	Selected data post-harvesting	Comments
Africa						
Australasia						
Baillie et al. (2005)	New Zealand (Northland)	Yellow-brown earths	100% and 25% of catchment	Increase in maximum stream temperatures, decrease in DO	Mean max. monthly temperatures increased by 5.6 °C (25% harvest) and 3.6 °C (100% harvest) Post-harvest DO 71% (25% harvest), 37% (100% harvest)	Response influenced by clear-cut harvest to the stream edge followed by stream-cleaning and stream width.
Bubb et al. (2002)	Australia (south-east Queensland)	Siliceous Sands, Lithosols, Grey and Gleyed Podzolics, Yellow Earths, Yellow Podzolics and Humus Podzols	100% of catchment	Similar TN, TP and SS concentrations at upstream and downstream monitoring sites indicate minimal impacts from harvest. TN, TP, SS loads episodic, reflecting the ephemeral nature of the streams	Streamflow-weighted concentrations (mg L^{-1}) TN: 0.36 to 2.01 TP: <0.01 to 0.28 SS: <10 to 264	Minimal impacts on water quality attributed to the retention of buffers 10–30 m wide, processing at the stump, retaining harvest residues on site, and use of site specific preparation practices to minimise erosion
Fahey and Marden (2006)	New Zealand, (Hawkes Bay)	Recent (Tephric and Orthic types) and Melanic soils	100%	Increased sediment yield	Sediment yield from harvesting (includes roading) 330 tonnes km^{-2} yr^{-1}	Sediment yields returned to pre-harvest levels within 2–3 years, attributed to good maintenance of infrastructure, oversowing and rapid replanting
Hopmans et al. (1987)	Australia (Victoria)	Red-brown loam	100% excluding a 30-m stream buffer	Changes in water quality were insignificant or minor but significant increase in sediment and nutrient exports with associated increased post-harvest discharge	Post-harvest export (kg ha^{-1}) SS: 50.85Cl: 20.55Na: 15.49 Mg: 10.39 Ca: 5.12 K: 3.95	Low impacts on water quality attributed to the retention of a 30 m buffer. Exports returned to pre-harvest levels within 18 months
Asia						
Douglas et al (1992)	Sabah, Malaysia		100% select logging	SS yields in harvest site increased 4-fold compared with control site after roading, 5-fold after roadside harvesting and 18-fold after catchment harvesting	Peak SS: 5734 mg L^{-1} well over 1000 mg L^{-1} for most storms, post-harvest	Based on storm event sampling, some degree of SS yield recovery one year after harvest
Ling et al. (2016)	Sarawak, Malaysia	Red-Yellow Podzolic soils	Unknown	Harvesting increased water temperature and TSS. TN, total ammoniacal-N, nitrate-N, TP and soluble reactive phosphorus were variable, no impacts identified. No identified impacts on pH and DO. Water quality (mainly turbidity, SS and nutrients) deteriorated significantly after rain, attributed to logging.	Logged sites: SS 10 to 16 mg L^{-1} Temp: > 25 °C Turbidity: ≈2.0 to 7.5 NTU	Variation in site characteristics likely obscured some of the harvesting effects.
Marryanna et al. (2007)	Selangor, Malaysia		100% clearfell, with 20 m buffer	Conductivity, turbidity increased after clearfelling; buffer strip mediated the effects of clearfelling.	with buffer/without buffer Mean Turb: 4.21 NTU/5.59 NTU Mean Conductivity 12.90 µS cm^{-1}/16.38 µS cm^{-1}	Water quality showed some improvement in the first two years after harvest as the vegetation recovered
Oda et al. (2011)	Chiba, Japan (Fukuroyamasawa)		100% of catchment	Decrease in baseflow Cl and increased nitrate concentrations after harvest. Storm event sampling, nitrate, K, Na, and Ca concentrations increased with discharge.	Peak NO_3: 280 µeq L^{-1} storm sampling 700 µeq L^{-1}	Highlighted the importance of sampling storm flows long with baseflows to assess harvest impacts on water quality
Tokuchi and Fukushima (2009)	Nara, Japan (Mt. Gomadan)		100% of each catchment	Nitrate concentrations significantly increased after clear-felling. Long-term, nitrate concentrations decreased with increasing stand age, whereas Ca, Mg, and Na increased. Ca and Mg concentrations were also strongly influenced by catchment characteristics (geology and topography).	NO_3: Max 057 $mmol_c$ L^{-1}	Used chronosequencing to assess the long-term influence of clear-cutting on water quality.

1 No studies were found on harvesting and water quality in Africa.

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