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Citation for published version:
Jozsa, J, Kiely, G & Borthwick, A 2014, 'Sediment Flux and Its Environmental Implications' Journal of environmental informatics, vol. 24, no. 2, pp. 111-120. DOI: 10.3808/jei.201400287

Digital Object Identifier (DOI):
10.3808/jei.201400287

Published In:
Journal of environmental informatics

Sediment Flux and Its Environmental Implications

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ABSTRACT

Sediment transport in fluvial systems is a key driver of basin-wide global soil loss, river sedimentation, and the movement and transformation of organic, inorganic, and nutrient materials, all of which can contribute to severe eco-environmental degradation. Since the late 1800s, much research effort has focused on the physics of sediment entrainment, transport, and deposition by river flows. This paper reviews ongoing research aimed at considering the simultaneous physical, chemical and biological processes that characterize riverine sediment flux. Four related issues are considered: riverine sediment flux; soil erosion and chemical transport; fluxes of dissolved organic carbon; and sediment-induced CO_2 emission/sequestration. Modelling of sediment flux has moved beyond empirical and statistical approaches to that of a generalized form of the universal integral solution of the basic flux equation, which is now anticipated to lead to a wide range of applications. Whereas soil erosion and riverine chemical transport are now known to cause soil degradation and reduced water quality, limited progress has been made to date on the quantification of erosion rates. As soils erode, CO_2 is emitted at erosion and transport sites and sequestered at deposition sites, the net effect being carbon sequestration. However, the rates of CO_2 emission/sequestration vary widely, owing to the large spatial variations in soil type, land-slope, rainfall intensity, etc. It is now well established that dissolved organic carbon (DOC) concentrations and fluxes have been increasing over the past two decades, due to reduced atmospheric sulphur concentration, climate warming, and changes in precipitation patterns. The research discussed herein provides insight into the interaction between sediment and multiple material substances, leading to a better understanding of fluvial river ecosystems, which is essential for maintaining river health.

Keywords

Sediment flux, soil erosion, environmental implication, chemical transport, dissolved organic carbon, CO_2 flux,

1. Preamble

Conventionally, the main surface material transport processes associated with sediment movement in a river basin may be categorized as runoff and sediment yield, soil erosion and nutrient loss, and river geomorphological evolution. The material transport processes tend to be considered by hydrologists, sedimentologists, hydraulic engineers, agronomists, ecologists, and environmentalists (Kisi et al., 2013; Van Rijn et al., 2013; Miller et al., 2014), whereas the geomorphic evolution of a watershed is the primary concern of geomorphologists (Provansal et al., 2014; Toone et al., 2014). In recent years, the environmental effects of sediment movement through a river basin have been investigated in terms of variations in natural organic matter, nutrients, and contaminants in water-sediment two-phase systems, extending to multiphase systems of water-sediment-carbon, water-sediment-nitrogen, and water-sediment-phosphorous, owing to the considerable annual losses of carbon, total nitrogen (TN) and total phosphorous (TP) to surface waters (Panagopoulos et al., 2007; Mora et al., 2014). At the time of writing, there is an increasing research focus on the migration and transformation of the organic and inorganic forms of carbon, largely because of their sensitivity to global environmental change (Raymond and Bauer, 2001; Galy et al., 2007). Obviously, a systematic description of the coupled processes of material transport in multiphase river systems is essential if we are to assess accurately the environmental impact of sediment transport.

The aim of this review paper is to provide an overview of the present state of knowledge of sediment flux and its environmental implications, thereby achieving a better understanding of river systems and their sustainability. The conventional focus has been on sediment generated from soil erosion in a basin with associated chemicals absorbed onto solids or dissolved in water. However, nutrients have become of major concern to environmentalists, while carbon is the primary issue in environmental, ecological and climatic study areas concerned with global change, in terms of both dissolved organic carbon and gaseous CO_2 flux. Hence, this paper discusses four inter-related issues closely related to sediment transport processes, including sediment flux, soil erosion and chemical transport, dissolved organic carbon, and sediment-induced CO_2 emission and sequestration. This approach has the advantage of treating the different facets of sediment transport in the spirit of a generalized integral form of the basic flux equation, which opens up a wide vista of future applications in earth science.

2. Sediment Flux

Water and sediment are the main carriers of various materials in river basins. The motion of the water-sediment mixture can have severe environmental consequences, owing to the sediment load carrying materials such as natural organic matter, nutrients and contaminants (Iwata et al., 2013; Thouvenot et al., 2007). Many theories have been proposed to describe the characteristics of sediment transport (Abad et al., 2008; Ganju and Schoellhamer, 2009; Zhang et al., 2013). Of these, the most frequently used theories are based on continuum concepts. The continuum assumption, which has proved very successful in representing liquid-gas motions, intuitively appears insufficient to describe motions of discrete solid particles in two-phase flows. Meanwhile, stochastic models can be used to model the motions of individual particles in a fluid. However, stochastic models assume that the random jump of each particle is independent and fits a Markov process, and so can only be applied to homogeneous turbulent flows and do not properly describe the interactions between solid particles. More sophisticated studies adopt a combined approach for describing solid-liquid systems whereby the liquid phase is modeled by means of the continuum concept and the solid phase using kinetic theory, with proper consideration taken of the interactions between the two phases.

The suspended sediment transport rate is usually calculated by integrating the product of the sediment velocity and concentration over the depth. In uniform flow in a simplified open channel, suspended sediment transport at equilibrium is reasonably well described by means of standard vertical profiles for the flow velocity and the concentration of suspended sediment. For engineering purposes, the velocity profile may be satisfactorily represented by a logarithmic distribution, provided the flow contains low concentrations of sediment (i.e. is dilute). However, this may not be the case for hyper-concentrated sediment-laden flow. Alternatively, the vertical sediment profile could be approximated by the well-established formula derived by Rouse (1937) from the principle of mass conservation. Due to the limitations of the Rouse formula, considerable research effort has gone into improved modeling of the concentration distribution of suspended sediment. This has led to diffusion, energy, mixture, similarity, stochastic, and two-phase flow theories, from which various formulas have been developed (see e.g. Rouse, 1937; Knapp, 1938; Bakhmeteff and Allan, 1946; Bagnold, 1962; Ananian and Gerbashian, 1965; Matalas, 1970; Drew, 1975; McTigue, 1981; Mendoza and Zhou, 1995). One of the most exciting advances made in the past few decades was by Ni and Wang (1991), who proved that a similar differential equation would be finally derived no matter which of the aforementioned theories was selected, which directly led to a generalized formula as an universal integral solution of the basic equation. Ni and his colleagues also demonstrated that most well-known formulas such as those proposed by Rouse (1937), Lane and Kalinske (1941), Hunt (1954), Ananian and Gerbashian (1965), Zagustin (1968), Laursen (1980), and Itakura and Kishi (1980) were merely special cases of the generalized formula under different conditions. This stimulated further studies which are still on-going (Cheng et al., 2013; Kundu and Ghoshal, 2014), the aim being to extend the general expression to an increasingly wide range of applicability.

Given that the suspended sediment transport rate is determined by the integral product of the sediment velocity and concentration over the flow depth, a plethora of formulas have been derived from different mathematical expressions for velocity and concentration profiles (Buyevich, 1990; Rasteiro et al., 1993; Davis and Gecol, 1994; Cheung et al., 1996; Xue and Sun, 2003; Deng et al., 2008; Bai and Duan, 2014). However, various further aspects must also be considered when calculating sediment transport in unsteady flow, in a non-straight channel or in a human disturbed river system (Lenzi and Marchi, 2000; Sammori et al., 2004; Francke et al., 2008; Marttila and Kløve, 2010; Gao and Puckett, 2012; Kabir et al., 2014). For example, although sediment-discharge hysteresis loops have been much analyzed in order to facilitate a better understanding of sediment transport processes, it remains unclear how to characterize accurately the hysteresis using indices (Aich et al., 2014). Moreover, to evaluate anthropogenic changes to river channels, full account must be taken of discontinuities in flow and sediment transport, and their effect on primary geomorphic parameters such as the active channel width, bed slope, and sediment grain size.

River water and sediment fluxes are closely related to runoff and sediment yield in a river basin. However, the descriptions of soil erosion used nowadays mainly derive from statistical and physical models based on causality. As a core activity of global change research, e.g. within the IGBP and IHDP programs, assessments of water and soil loss are made at three spatial scales; namely, hillside, watershed, and regional scales. Statistical models focus on the establishment of empirical relationships between water, soil loss, and various influencing factors; of such models, the most widely used include the universal soil loss equation (USLE) (Wischmeier, 1976) and the revised universal soil loss equation (RUSLE) (Renard et al., 1991). Physical models are usually based on deterministic theories for hydrodynamics and sediment transport, which are used to predict runoff and sediment yield in small basins. Examples of physical-deterministic models include CREAMS (Chemicals, Runoff and Erosion from Agricultural Management Systems), GLEAMS (Groundwater Loading Effect of Agricultural Management Systems), CSEP (Climate Index for Soil Erosion Potential), EPIC (Erosion-Productivity Impact Calculator), ANSWERS (Areal Nonpoint Source Watershed Environment Response Simulation), AGNPS (Agricultural Nonpoint Pollution Source), KINEROS (Kinematic Runoff and Erosion Model), MEDALUS (Mediterranean Desertification and Land Use), EUROSEM (European Soil Erosion Model), WEPP (Water Erosion Prediction Project) (Laflen et al., 1991) and LISEM (Limburg Soil Erosion Model). Of these, CREAMS (Knisel, 1980) establishes a field model primarily applicable at a scale incorporating hill-slope and valley bottom (Rudra et al., 1998), which partially meets requirements for the protection of water resources. At watershed-scale, ANSWERS (Beasley et al., 1980) considers the relationship between non-point source pollution and soil erosion. The individual rain distributed models LIMSEM (De Roo, 1996) and EUROSEM (Morgan et al., 1998) are both physically-based. Nevertheless, the mechanism model WEPP (Laflen et al., 1991) released by USDA in 1995 accounts for rainfall infiltration, irrigation, surface runoff, soil separation, sediment transport and deposition process, plant growth and decomposition of residues, and is by far the most complicated calculation model for the prediction of soil erosion. In recent years, several new models based on stochastic theory (Chen et al., 2013; Foufoula-Georgiou and Stark, 2010) and self-organization concept have been applied to the simulation of slope surface erosion (Han et al., 2011) and have the potential to predict rill evolution in detail.

The accurate prediction of water and sediment variations at different temporal and spatial scales is very difficult using the aforementioned models owing to excessively high data requirements and limitations of scale-up. Most conventional methodologies for soil-erosion assessment are limited to small or medium river basins. Although efforts have been made to develop an alternative approach for soil-erosion intensity assessment in large basins (see e.g. Ni et al., 2014), more information about the influencing factors is still needed from systematic field studies.

3. Sediment and Soil Chemical Transport Processes

Migration and transformation of soil organic matter and other chemical nutrients occur simultaneously with soil erosion and sediment transport. Such processes can result in decline in soil fertility, reduction in crop yield, and release of chemical components into rivers, lakes or reservoirs, perhaps leading to non-point source pollution and eutrophication. It is generally believed that nutrient loss from surface soil is driven by nutrients becoming dissolved by runoff and/or being carried away through sediment transport. Hitherto, nutrient loss from slope surface soil has mostly been determined from rainfall-runoff plots through real-time monitoring and analysis of runoff, sediment and nutrient. For example, a five-year continuous field observation on the loss of nitrogen due to growth season drainage in the United States, showed that losses by use in the case of fertilizers were 48.8 kg, 96 kg and 144 kg nitrogen per hectare, corresponding to 4.8, 9.6 and 12.7 times that obtained for no fertilizer (Almasri and Kaluarachchi, 2004).

Water and sediment are not only the main carriers of other materials but also affect their migration and transformation. During soil erosion and sediment transport, the flow of water and sediment influences the dissolution of inorganic and organic components in sediments as well as the redistribution of external contaminants between solid and liquid phases, e.g. adsorption and desorption (McCulloch et al., 2003). These can further alter sediment composition and the occurrences of pollutant contamination between the different phases, ultimately affecting the water environment status. Moreover, inorganic components in soil or sediment could also affect the retention of metal species, and further affect their migration behavior through water and sediment transport. Background values of trace elements in sediments directly determine the species and content of various background ions in the water phase, altering the adsorption of organic pollutants onto sediment. Humic organic components in soil or sediment also play an important role in interphase distribution or dissolution, affecting not only water quality but also the adsorption and desorption of organic matter (Grathwohl, 1990; Kile et al., 1995; Luthy et al., 1997).

In flowing water, the presence of sediment also affects biodegradation and photolysis of organic pollutants, leading to their transformation between either liquid and gas or solid and gas phases. It has been reported that sediment can promote the biodegradation of organic pollutants (Xia and Wang, 2008; Duong et al., 2009). Due to the enrichment of various nutrients in sediment particles, the sediment itself provides better conditions for microbial growth. Sediment acts as a carrier for microbial adhesion and metabolic activity, thus encouraging rapid proliferation of microorganisms. Moreover, sediment also transports pollutants and provides direct contact conditions beneficial to microorganisms and organic matter (Marchesi et al., 1994).

Although the degradation of organic matter has been a focus for interaction between water, sediment and pollutants in terms of its impact on water quality and pollutant migration and transformation, such degradation does not necessarily mean complete detoxication and mineralization. For example, organic pollutants such as steroids are by no means completely removed in the degradation process. Instead, they are transformed into other intermediates maintaining a potential ecological hazard; in this context, the presence of sediment may promote further transformation of pollutants. Mineralization (involving CO_2 emission) is of key importance in understanding the mechanisms behind migration and transformation of organics in sediment-laden flow, and could serve as a useful indicator for environmental and ecological consequence diagnoses.

4. Sediment-induced CO_2 emission and sequestration

During sediment movement, mineral weathering is most active (Lal, 2003; Berhe et al., 2007), with physical, chemical and biological processes relevant to organic carbon decomposition, synthesis and transformation also affected, leading to CO_2 sequestration, emission, and change of concentration in the atmosphere (Stallard, 1998; Smith et al., 2001; Berhe et al., 2007; Quinton et al., 2010).

The three main pathways for CO_2 exchange between soil and atmosphere include chemical weathering of minerals in the soil, and the formation and decomposition of soil organic matter (Suchet and Probst, 1995; Van Oost et al., 2007). CO_2 fixation of inorganic minerals in sediment-laden flow is usually attributed to accelerate chemical weathering via sediment transport. Silicate and carbonate in broken soil particles generate soluble bicarbonate by absorbing CO_2 as runoff-induced scouring occurs, creating a "carbon sink" (Meybeck, 1982; Berner et al., 1983; Gaillardet et al., 1999). Although the consensus is that the processes of CO_2 sequestration and emission of organic carbon in sediment-laden flow are primarily caused by changes to the behavior of soil organic carbon at different stages of sediment movement, controversy still exists about the exact effect of CO_2 emission or sequestration at a specific stage (e.g. erosion, transport or deposition).

In a region of sediment erosion, soil particles undergoing crushing and migration cause organic carbon decomposition to speed up, thus releasing more CO_2 (Lal, 1995; Lal, 2003; Jenerette and Lal, 2007). Meanwhile, the loss of soil organic carbon during soil erosion helps reduce surface plant growth due to diminished fertility and volume of residues available from fields, resulting in loss of the soil carbon pool (Lal, 1995). An alternative view is that soil formation and CO_2 fixation are accelerated due to surface organic carbon loss in an erosion area, resulting in the so-called "substitution effect" (replacement) thereby increasing the storage of the soil carbon pool (Stallard, 1998; Harden et al., 1999; Smith et al., 2001; Harden et al., 2002; McCarty and Ritchie, 2002; Fontaine et al., 2007; Quine and Van Oost, 2007; Van Oost et al., 2007;

Quinton et al., 2010).

Soil organic carbon is believed to be further decomposed in areas dominated by sediment transport, though different views exist regarding the decomposition ratio. Smith et al. (2001) used an equilibrium model to point out that the longitudinal decomposition of organic carbon is almost negligible. However, Jacinthe and Lal (2001) found, by interpreting experimental data, that approximately 15% of soil organic carbon was converted to CO_2 in the sediment transport process. Óskarsson et al. (2004) argued that the organic carbon decomposition rate could reach 50% in the process, and Schlesinger (1995) reckoned that 100% soil organic carbon could transform into CO_2.

Harden et al. (2002) suggested that upstream sediment entering areas of sediment deposition tends to enrich organic carbon, increasing CO_2 decomposition and emission. Other investigators (Stallard, 1998; Berhe et al., 2007) have claimed that a protective layer would form on the original soil due to sediment deposition, hindering subsoil decomposition and reducing the rate of release of CO_2. However, the sediment organic carbon content in the protective layer is often lower than that of the carbon-rich original surface soil, and thus the soil balance once again breaks, accelerating CO_2 fixation and enhancing storage of the carbon pool (Stallard, 1998; Smith et al., 2001).

In certain countries, including Bangladesh, Brazil, Burma, China, India and the USA, the sediment content of major rivers can be relatively high (e.g. the Amazon, Yellow and Ganges-Brahmaputra all carry a mean sediment load of ~10^9 tons per annum; see e.g. Milliman and Meade, 1983; Goodbred and Kuehl, 2000), and so the process of material fluxes becomes more complicated. For example, substantially different chemical and biological behaviors of inorganic/organic carbon and organic pollutants have been reported for Chinese rivers with high sediment content (Marshall et al., 2000; Xia and Wang, 2008; Duong et al., 2009). To date, almost no research studies have considered the environmental consequences of sediment-laden river flows in terms of CO_2 emission and sequestration. This is likely to provide fertile ground for future scientific studies.

Through interaction with carbonate rocks, biophysical and biochemical processes play an important role in the global carbon balance. Biological metabolic processes involve carbonate activation. Biological composition, structure and activity affect (directly or indirectly) the circulation and transformation of soil organic carbon throughout the whole process of sediment movement. Soil microorganisms provide a constantly updated dynamic driver for soil carbon form transfer, which continuously assimilates materials in the environment as part of the microorganisms' metabolic processes while releasing carbon components in different forms to the environment. Until now, there has been no unified understanding about biological effects on the soil carbon pool in sediment erosion regions. In such regions, biological photosynthesis and respiration also cause the content of inorganic carbon in river water to vary, although this may be subject to the influence of sediment content. Moreover, sediment, as a carrier of pollutants and microorganisms, could cause changes to the biological behavior of organic pollutants in water (Marshall et al., 2000; Duong et al., 2009), hence altering rates of CO_2 emission and sequestration. In depositional areas, the biological mineralization of soil organic carbon becomes a very complicated process affecting a wide range of parameters, including soil properties, temperature, moisture, and organic carbon composition.

Different degradation rates of soil organic carbon are likely in areas which experience different erosional, transport and depositional processes associated with sediment movement (Berhe, 2012). Obviously, considerable attention should be paid to the influence of biological processes on the carbon cycle, in the overall context of sediment processes.

5. Dissolved Organic Carbon (DOC)

In addition to the vertical pathway of exchange of carbon (i.e. CO_2) between terrestrial ecosystems and the atmosphere, there is also a horizontal pathway of significant soil carbon loss of dissolved organic carbon (DOC) to the riverine environment. The hydrological erosion pathway is more by sub-surface flow than by surface runoff, and as such the riverine concentrations of DOC are a function of myriad factors, including climate, season, soil type, ecosystem type, temperature, rainfall, and antecedent soil moisture. While the concentration of DOC is in itself an informant variable, the flux of DOC (i.e. the product of DOC concentration and riverine flow rate) provides greater insight into DOC impact on carbon loss and water quality. The problems associated with DOC in riverine water are three-fold: firstly its carbon loss from soils contributes to soil quality degradation; secondly, as DOC export is a source term in catchment carbon budgets, increasing DOC loss (export) may result in some catchments (especially peatlands) becoming sources for carbon and thus destabilizing their large soil stores of carbon; and thirdly its negative impact on water quality if high DOC waters are used as the raw water in potable water treatment plants and then treated with chlorine disinfection possibly resulting in elevated levels of carcinogenic trihalomethanes (THMs) and other toxins.

A number of studies have shown that DOC riverine concentrations have been increasing over the past two to three decades, especially over Northern Europe and North America (Filella and Rodriguez-Murillo, 2014; Monteith et al., 2007; Evans et al., 2005; Grieve and Gilvear, 2008; Mehring et al., 2013; Tian et al., 2013; Oni et al., 2014; Sucker and Krause, 2010). Several hypotheses have been tendered as possible explanations for this DOC increase, including: decreasing atmospheric sulphur concentration; climate warming (with seasonal temperature increases); increasing precipitation with increasing annual (e.g. winter/spring) river discharge; reducing summer discharge; longer inter-annual drought length; increasing atmospheric CO_2 concentration; CO_2 mediated stimulation of primary productivity; increasing decomposition; land-use land-management change (e.g. afforestation of peatlands; wind farm developments and disturbance); catchment scale; and climate zones.

Freeman et al. (2004) in peatland manipulation experiments found that reduced summer precipitation did not explain increases in DOC concentration. Noting increases in atmospheric CO_2 (CO_2 enrichment), Freeman et al. proposed that DOC increases were induced by increased primary production and DOC exudation from plants. Evans et al. (2005), in a study of 22 UK upland waters, found that DOC concentrations increased by an average of 91% during the previous 15 years and noted that this increase resulted from a combination of declining acid rain deposition (reducing atmospheric sulphur concentration) and rising temperatures. In an assessment of data from 522 remote lakes and streams in North America and Northern

Europe, Monteith et al. (2007) found that DOC concentrations increased in proportion to the declining rates of atmospherically-deposited anthropogenic sulphur. Monteith et al. stressed that the rise in DOC concentration was integral to the recovery from acidification. Grieve and Gilvear (2008) in a study of tributaries (disturbed due to wind farm construction on blanket peatlands versus undisturbed) found DOC concentrations to be always higher in the disturbed streams by concentrations ranging between 2 and 5 mg/l. A review by Sucker and Krause (2010) found that the most realistic reason for DOC increases was the complex interaction of changing atmospheric sulphur deposition and climate warming. Clark et al. (2010) observed a stalemate had occurred in the debate as to why DOC increases, between those favoring decreasing atmospheric sulphur deposition and those supporting climate warming. Clark et al. suggested that the conflicting observations may be due to them being derived from experiments taken at different spatial and temporal scales.

In a review of DOC cycling and transformation in riverine and estuarine waters, Bauer and Bianchi (2011) noted that DOC is derived from terrestrial vegetation and soils. Bauer and Bianchi observed that estuaries bordering the Gulf of Mexico have among the highest DOC concentrations as well as the some of the highest rates of fresh litter decomposition. Kindler et al. (2011) found that DOC losses constitute a small but continuous loss of carbon from terrestrial ecosystems, typically of the order of 25% of net ecosystem exchange and as such must be incorporated in carbon budgets. Laudon et al. (2012) in a study of 49 catchments in Northern latitudes found that the mean annual temperature (MAT), in the range of -3 to 10 ^{o}C, has a strong control over regional stream water DOC concentration, with the highest concentrations in regions with mean annual temperature ranging between 0 and 3 ^{o}C.

Räike et al. (2012) examined 36 years of data from Finland, and reported increases in DOC stream water concentrations but no increase in DOC export. This holds when precipitation and stream flow decreases, possibly due to climate warming. Oni et al. (2013) studied three nested headwater boreal catchments, and found that stream DOC was positively-correlated with certain trace metals (copper, iron and zinc) and negatively-correlated with several other chemical parameters (sulphate, conductivity, and calcium). These observations indicate the subtle effects of recovery from acidification. However, Oni et al. (2013) concluded that climate warming rather than recovery from acidification could be the dominant driver of DOC increases in the boreal catchments they considered. Mehring et al. (2013) found that long drought periods in North American rivers reduced DOC concentration (in summers) followed by higher DOC concentration in the later hydroperiod (autumn/winter). Tian et al. (2013) observed that a linear relationship held between the surface temperature and mean in-stream DOC concentration at the annual scale for seven major watersheds, including coastal rivers crossing different climate zones. Tian et al.'s results strongly suggest that climate warming is the primary factor causing the increasing DOC flux. Tian et al. also note that landscape factors are a secondary consideration.

In a recent study, Koehler et al. (2009) found the concentrations of DOC in peatland stream water in Ireland, ranged from 2.7 mg/L to 11.5 mg/L over one year with the higher concentrations in the summer. The DOC concentrations were highly correlated with temperature. However as the flow rates were much higher in winter, the export of DOC was highest in winter. The annual export of DOC for the year 2007 was 14.1 g $cm^2ha^{-1}yr^{-1}$. This was approximately twice that of the carbon in CH_4 emissions and approximately half that of the atmospheric carbon sequestered by the peat soils.

Liu et al. (2014) investigated the spatial and seasonal variation of DOC concentrations in 55 Irish streams on seven time occasions over 1 year (2006/2007). The DOC concentrations ranged from 0.9 to 25.9 mg/L with a mean value of 6.8 and a median value of 5.7 mg/L and varied significantly over the course of the year. The DOC concentrations from late winter (February: 5.2 ± 3.0 mg/L across 55 sites) and early spring (April: 4.5 ± 3.5 mg/L) had significantly lower DOC concentrations than autumn (October: mean 8.3 ± 5.6 mg/L) and early winter (December: 8.3 ± 5.1 mg/L). Stream runoff from peat soils had the highest DOC concentrations and the highest DOC export while the lowest were from catchments with mineral soils (with soil organic matter (SOM) < 3%). The DOC production sources (e.g., litterfall) or the accumulation of DOC over dry periods might be the driving factor of seasonal change in Irish stream DOC concentrations. Analysis of data using stepwise multiple linear regression techniques identified the topographic index (TI, an indication of saturation-excess runoff potential) and soil conditions (organic carbon content and soil drainage characteristics) as key factors in controlling DOC spatial variation in different seasons. The TI and soil carbon content (e.g., soil organic carbon; peat occurrence) are positively related to DOC concentrations, while well-drained soils are related to DOC concentrations. Similar observations have been noted by Worral et al. (2006), Worral and Burt (2007), Eimers et al. (2008), Dawson et al. (2008), and others.

There is growing concern internationally amongst water treatment plant (WTP) operators and water quality (WQ) regulatory agencies with regard to the levels of natural organic matter (NOM) such as DOC present in raw water and persisting in treated water prior to disinfection. The presence of elevated NOM can cause problems in water treatment processes at drinking WTPs and problematic WQ of water when treated with chlorine disinfectant. Problems occur not just as the treated water leaves the WTP but also along the distribution network and more crucially at the consumer's tap. The problems at the WTP include: negative WQ effects on aesthetics, colour, taste and odour; inefficiencies in coagulation/flocculation processes leading to smaller floc sizes and more expensive floc settlement costs; the requirement for activated carbon process (GAC) and the production of elevated levels of trihalomethanes (THMs), haloacetic acids (HAAs) and other toxins in the drinking water (EPA and HSE, 2011; EPA, 2012). Elevated NOMs can lead to the promotion of an unhealthy biological growth in the water distribution network. At the tap, the above issues are integrated, resulting in: poor WQ in colour, taste and odour (including chlorine odour); and elevated THMs and other toxins. Chlorine has a long track record (more than 100 years; USEPA, 1999) of success. THMs are a group of organic chemicals which are considered to be carcinogenic in excessive amounts. EU regulations have an upper limit of 100μg/L for total THMs, which are composed of the four compounds: Chloroform, Bromoform, Dibromochoromethane, and Bromodichloromethane. Haloacetic acids (HAAs) are a further group of chlorine associated DBPs, receiving more recent attention. The higher the chlorine dose, the higher the THMs (Kraus et al., 2010). The levels of NOM concentration have significant variation on the temporal and spatial scales. Typically NOMs increase in flood events and decrease in low flow periods. While the high river flow seasons of Autumn and Winter are more likely to have highest NOM concentrations, periods of NOM flushing can occur in Spring and Summer flood events (after dry periods). NOMs are thus considered to vary over the seasons and even from year to year, depending on the climate. NOMs are also known to increase with temperature (Koehler et al., 2009) and possibly with climate change. Upland peatland catchments tend to have high NOMs which tend to be diluted in the downstream direction as the catchment size enlarges. However, where rivers flow into lakes, and lakes are used as raw water sources for WTPs, then the low velocity lakes can retain elevated NOM concentrations. Understanding both chemical and physical characteristics of NOM in source waters is key to better water treatment (Wei et al., 2008).

6. Conclusions

Modeling of runoff and sediment yield in multiple-scale watersheds remains a challenging problem even though considerable progress has been made on understanding the mechanics of sediment transport from entrainment to deposition. However, the present review highlights new problems that are emerging about the effects of sediment motion, noting increasing environmental and ecological concerns at scales from hillside to global. In a river basin, it is necessary to consider the integrated physical, chemical and biological aspects of sediment flux in order to appreciate the wider impact on the eco-system. The review has shown that it is necessary to consider simultaneously soil erosion, sediment transport and the associated movement of dissolved organic carbon and chemicals, along with horizontal and vertical carbon exchanges. This leads naturally to the concept of a universal flux equation that integrates all the foregoing aspects of sediment flux. More field data are required on soil erosion and carbon exchanges at different spatial scales. Future research effort needs to be directed towards a better understanding of integrated sediment transport processes in multi-phase systems, and their environmental consequences. Our understanding of the biological response to sediment flux needs strengthening particularly in the context of carbon and nitrogen transformations, including sequestration and emission of greenhouse gases accompanied with sediment erosion, transport and deposition. This can only be achieved by a combination of fundamental laboratory-based research into soil-water-sediment science and high quality field observation campaigns conducted in major river basins at sufficient spatial resolution. A relevant example of the former is provided by Wang et al. (2014) who recently measured the soil organic carbon, dissolved organic carbon and CO_2 fluxes in a laboratory-scale flume containing loess soil subjected to simulated rainfall.

Acknowledgements

The third-named author undertook part of this study while working on an EU FP7 Marie Curie Actions Project PIRSES-GA-2011-294976 on Geohazards.

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