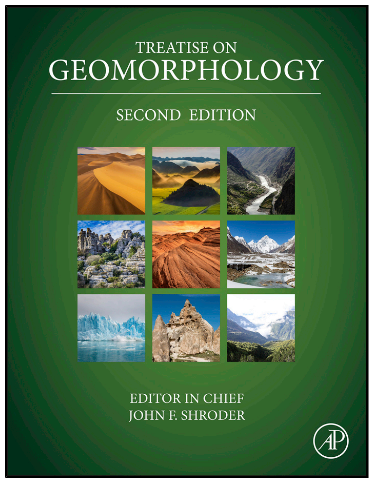

From James, L.A., Lecce, S.A., Pavlowsky, R.T., 2022. Impacts of Land-Use and Land-Cover Change on River Systems. In: Shroder, J.J.F. (Ed.), Treatise on Geomorphology, vol. 6. Elsevier, Academic Press, pp. 1191–1236. https://dx.doi.org/10.1016/B978-0-12-818234-5.00089-4.
ISBN: 9780128182345

6.52 Impacts of Land-Use and Land-Cover Change on River Systems

L.A. James[a], **S.A. Lecce**[b], **and R.T. Pavlowsky**[c], [a] University of South Carolina, Columbia, SC, United States; [b] East Carolina University, Greenville, NC, United States; and [c] Missouri State University, Springfield, MO, United States

This is an update of L.A. James, S.A. Lecce, 9.37 Impacts of Land-Use and Land-Cover Change on River Systems, edited by John F. Shroder, Elsevier Inc, 2013, https://doi.org/10.1016/B978-0-12-374739-6.00264-5.

6.52.1 Introduction

The extent to which humans have modified fluvial systems through changes in land use is difficult to overestimate. Changes in land use have impacted the delivery of water, sediment, nutrients, contaminants, and other materials downstream, and can alter water and sediment quality, aquatic habitats, and channel and floodplain morphology over short to intermediate time-scales (10^{-1}–10^{-3} year). To some extent, this was known to Greek philosophers (Glacken, 1967), and it has been known to

Treatise on Geomorphology, Second Edition, 2022, 1191–1236

modern science since George Perkins Marsh (1864) described the decreased permeability of Earth and the erosive effectiveness of runoff:

> The face of the earth is no longer a sponge, but a dust heap, and the floods which the waters of the sky pour over it hurry swiftly along its slopes, carrying in suspension vast quantities of earthy particles which increase the abrading power and mechanical force of the current, and, augmented by the sand and gravel of falling banks, fill the beds of the streams, divert them into new channels and obstruct their outlets.
>
> (Marsh, 1864: 215).

Various topics of land-change science have been covered by several books (Turner et al., 1990; Anderson et al., 2013; Goudie, 2019), symposia proceedings (Thomas, 1956; Brierley and Stankoviansky, 2002, 2003; James and Marcus, 2006), and review papers (Turner et al., 2007; Downs and Piegay, 2019; Goudie, 2020). Where land-use change increases runoff or sediment supplies, flooding and sedimentation can increase and stream habitats can be severely altered (Jacobson et al., 2001). Where past land uses have generated a large sediment pulse, the storage of sediment in the system may explain modern patterns of sediment remobilization (Fryirs and Brierley, 1999; Brierley, 2010; Lyons et al., 2015; Sims and Rutherfurd, 2017). Legacy-sediment deposits associated with human activities can also contain nutrients or toxic pollutants (Miller and Orbock Miller, 2007; Pavlowsky et al., 2017). Modern and historical land uses in watersheds should be understood to develop a spatially distributed watershed perspective of the river and a sense of the temporal dynamics of the system. These are important considerations for river rehabilitation and management (Wohl et al., 2015).

Land use generally refers to Earth's terrestrial surface as modified by human activities, whereas land cover may refer to natural biophysical attributes of Earth's surface. This chapter addresses the indirect impacts of upland and floodplain land use on fluvial systems, but not direct alterations within river channels. Human impacts on rivers have been classified as '*direct changes*' within channels or to riparian vegetation versus '*indirect changes*' to land use, such as vegetation changes, agriculture, urbanization, and mining (Brookes, 1994; Brierley and Fryirs, 2005). A broad view is taken here by considering changes in land use and land cover (LU/LC) that may be driven by either human alterations or natural processes such as spatial and temporal variability in climate. Although the emphasis is on the effects of human activities through land-use change, isolating those effects from natural processes of land-cover change or from the effects of climate change is not always feasible. In many ways, watershed responses to anthropogenic and climate changes are similar, and evaluations of changes in LU/LC are logically covered in tandem with climate.

Separating the importance of anthropogenic and non-anthropogenic (i.e., climate, geomorphology, lithology, tectonic activity, etc.) factors in explaining change in fluvial systems (e.g., sediment yields) can be extremely difficult and results vary with spatial and temporal scales (Vanmaercke et al., 2015). For instance, Foster et al. (2003) concluded that sediment production rates in a small watershed in France were dominated by land use over the late Holocene but that climate change and meteorological events become more important over shorter timescales. Notebaert and Verstraeten (2010) argued that the linkage between sedimentation history and driving forces are based primarily on synchronicity rather than evidence for a direct causal relationship. Moreover, the question of 'What is natural?' arises with attempts to isolate purely human influences (Graf, 1996b; Wohl, 2001; Newson and Large, 2006; Wohl and Merritts, 2007; Montgomery, 2008; Fryirs and Brierley, 2009; Brown et al., 2018). If the subtle effects of Paleolithic hominids are considered to be a part of natural processes, then when do human impacts cease to be natural? Furthermore, accelerated human activity coincides largely with substantial mid- to late-Holocene climatic adjustments, so it is difficult to identify watersheds where either human activities or climate change did not occur, which are needed as controls for testing. These topics are explored in Volume 13 of this treatise, which focuses on human impacts and climate change. This chapter examines how rivers are influenced by changes in human LU/LC.

Anthropogenic transformations of the land not only have repercussions on fluvial systems but also have impacts on many other systems. Land-use changes exacerbate climate change (Houghton, 1995; Bonan, 1997; Pielke et al., 2011), cause species extinctions (Pimm and Raven, 2000; Murphy and Romanuk, 2014), introduce toxic pollutants to the environment, increase the production and delivery of runoff, sediment, nutrients, and other pollutants to rivers (Poff et al., 2006), and generate geomorphic responses (Brierley and Stankoviansky, 2003; James, 2019). Land-use change poses a growing challenge for the coming millennium with consequences that may exceed those of climate change (Sala et al., 2000; Vörösmarty et al., 2000; DeFries and Eshleman, 2004). Unfortunately, the impacts of anthropogenic land-use change on hydrologic systems have received far less attention than the impacts of climate change (Lambin et al., 2002; Brisset et al., 2017; Rogger et al., 2017), but a growing movement in land-change science started to emerge by the advent of the new millennium (Turner II et al., 2003, 2007; DeFries and Eshleman, 2004).

Factors driving global environmental change can be divided into four categories: hydroclimatic, sea-level rise, topographic relief, and human activity (Slaymaker et al., 2009). Such a categorization isolates human agency, which, outside of high-latitude regions, is the most effective and most rapidly varying cause of change. In Europe, the magnitude of human impacts over the past 2000 years has exceeded those of climate change (De Moor et al., 2008; Hoffmann et al., 2009; Notebaert and Verstraeten, 2010), and similarly over the past 200 years in many locations including fluvial sedimentation in the UK (Walling et al., 2003b) and the upper Midwest of the USA (Knox, 2006).

This chapter begins with a discussion of the factors that drive fluvial responses. This includes landscape sensitivity; that is, what landscapes are sensitive to change, their resilience, differences between upland changes that propagate downstream to rivers, and sensitivity to change by rivers themselves. It also addresses how spatial and temporal scales are linked in such a way that broad global-scale studies may require a historical perspective. The effects of climate change change on fluvial systems are briefly examined. Section 6.52.3 covers changes to flow regimes, soil erosion, and sediment yields that may result from land-use changes such as those caused by agriculture, urbanization, or climate change. Section 6.52.4 focuses on downstream responses of fluvial systems to the hydrological changes upstream. Rivers may respond to changes in upland water and sediment production by experiencing channel and floodplain metamorphosis through a variety of forms, at gradual or punctuated rates. River response depends on the nature of change to inputs and the character of the fluvial conveyance system. The emphasis of this chapter is on upland land uses and the responses of channels and floodplains downstream. Changes in floodplain land uses, such as vegetation clearance, are covered only peripherally. In-channel and floodplain changes, such as weirs, bypasses, channelization, levees, bank protection, riparian vegetation clearance, ditching, leveling, wetland drainage, and chemical applications, are not covered. The effects of dams and flow regulation are covered elsewhere in this volume and are not repeated here. Section 6.52.5 presents an overview of the history of human activities with an emphasis on the advent and spread of agriculture during the Neolithic, the diffusion and intensification of agricultural technology into Northern and Western Europe, its relatively rapid introduction into the Americas and Australia, resultant alluvial episodes, and the present state of legacy sediment stored in affected watersheds. In some locations in the New World, contrary to common assumptions, pre-European agriculture was geomorphically effective, but this remains to be tested by stratigraphic evidence throughout the Americas and Oceania. Similarly, episodic soil erosion and sedimentation following European colonization were not universal, and the hypothesis of rapid post-colonial alluviation should also be tested.

6.52.2 Factors governing geomorphic responses to land-use change

Three general factors should be addressed before considering specific geomorphic processes and consequences of land-use change. The sensitivity of systems to change and the spatial and temporal scales of change have a great bearing on the patterns that emerge. Superimposed on these two factors are the effects of climate change that may not be easy to separate from the effects of land-use changes. Recognizing these factors is essential to a full understanding of interactions between land use and responses of fluvial systems.

6.52.2.1 Landscape sensitivity

The likelihood that LU/LC changes will generate substantial responses in runoff or sediment production varies greatly between watersheds and through time depending on a variety of factors. A given change in one watershed may promulgate substantial responses, whereas the same changes in another watershed may have no effect. Moreover, the effectiveness of changes in water and sediment loadings in generating geomorphic responses downstream in rivers and floodplains may also vary greatly within and between basins (Fryirs et al., 2009). To place geomorphic impacts of land use into a proper perspective, contemporary sensitivities to geomorphic change should be considered. The concept of landscape sensitivity expresses variability in geomorphic response to change and can improve explanations and predictions of the potential magnitude, frequency, and location of responses to external influences, including both natural and human perturbations (Allison and Thomas, 1993; Notebaert and Verstraeten, 2010). This involves the tendency for change and the ability of the system to absorb changes (Fryirs, 2017) as well as rates of change (Reid and Brierley, 2015). The concept is important for communicating geomorphic risk among geomorphologists and between them and industry, river engineers, and the scientific community (Fryirs, 2017). Landscape sensitivity may increase with human activities such as agriculture or deforestation because it measures the susceptibility to change by frequently occurring events. Although definitions vary, this chapter adopts the definition expressed by Brunsden and Thornes (1979) as the likelihood that changes in controls of a system will produce a 'sensible, recognizable and persistent response' (cf. Brunsden, 2001). They described sensitivity as reliant upon the resisting elements of the landscape (barriers) and disturbing forces, both of which have strong spatial and temporal variabilities (cf. Brunsden, 1993). The concept includes at least three definitions (Allison and Thomas, 1993):

1. spatial variation in the ability of landforms to change (Brunsden, 1990),
2. susceptibility to disturbance or fragility (Huggett, 1988), and.
3. ability of landforms to resist change (Brunsden and Thornes, 1979).

The third definition may be regarded as a type of landscape resilience that has an opposite or inverse connotation (insensitivity). Resilience may also include recovery from change. Sensitivity implies an element of instability in the system that may result in rapid, irreversible change (Thomas, 2001). It varies with scale and is complicated by an intimate dependency on thresholds of geomorphic stability that require consideration of the magnitude and frequency of events (Thomas, 2001). Landscape sensitivity is generally taken to mean susceptibility to frequent, moderate-magnitude events, but geomorphic changes to insensitive landscapes may be extensive and enduring in response to large infrequent events (Evans, 1993). Note that the concept applies across a wide range of scales from large basins to individual channel banks.

Sensitivity of channels and floodplains is highly variable and depends not only on geomorphic and structural elements but also on their interactions and spatial patterns (Fryirs and Brierley, 2009). Antecedent conditions may also be important to river sensitivity and have been characterized in terms of geologic, climatic, and anthropogenic memories that influence landscape processes (Brierley, 2010). Land-use change is often associated with a substantial increase in sensitivity. For example—prior to European settlement—many North American landscapes were relatively insensitive to change owing to thick forest cover and modest pre-European human alterations relative to their Old World counterparts. Intense land clearance, tilling, and grazing practices introduced by European agriculture, however, greatly reduced resilience and increased geomorphic sensitivity and generated severe erosion and sedimentation in some cases.

Most studies of river metamorphosis have focused on areas of dramatic change, but those may be atypical. In the upper Hunters watershed of southeastern Australia, Fryirs et al. (2009) found that less than 20% of the channel length experienced metamorphosis, instead they found that transitions with a continuum of changes existed between stable and changed reaches. Many studies of landscape sensitivity have been concerned with perturbations to uplands or soil erosion (Boardman, 1993; Evans, 1993; Quine and Walling, 1993). The concept of landscape sensitivity extends beyond the sensitivity of the landscape that is hydrologically contributing (the upper watershed), however, to also include the sensitivity of geomorphic systems in receiving areas such as channels, fans, deltas, and floodplains. The inclusion of depositional areas is in keeping with definitions based on the likelihood that changes will produce a sensible, recognizable, and persistent response. Several applications of the concept have been made to fluvial systems (Downs and Gregory, 1993) and to interactions between upland systems and fluvial systems. Channel morphology and floodplain sedimentation rates may be linked to changes in the sensitivity of uplands to runoff generation and soil erosion. For example, Knox (2001) concluded that the introduction of agriculture in the upper Mississippi Valley during European colonization increased landscape sensitivity through reductions in infiltration and increased runoff generation and flood magnitudes that resulted in tributary channel enlargement. Based on ^{14}C inventories in eastern Mediterranean watersheds, Dusar et al. (2012) concluded that initial human land-use changes generated high rates of sediment production, which decreased when hillslope erosion left thin soils insensitive to erosive forces.

6.52.2.2 Scales of space and time

Careful evaluations of river responses to land-use change inevitably bring up questions of scale. Responses to LU/LC change vary with spatial scale. Small perturbations that are effective locally may be imperceptible or missing in larger systems at a broader scale. For example, early human impacts in the Rhine Basin were substantial in certain small watersheds but negligible in larger rivers until the nineteenth century (Lang et al., 2003). Temporal scales are also important. At Quaternary time scales Milankovic cycles may be involved and intermittent storage, which can dominate modern sediment budgets, is of less importance (Hinderer, 2012). Although the traditional focus on fluvial change has been on catastrophic (rapid) responses, the concept extends to protracted, gradual change (Allison and Thomas, 1993). For example, periods of subtly increased but persistent erosion and sedimentation during the late-Holocene may appear in the geologic record as a sudden episode of substantial sedimentation.

In a general sense, spatial scale is a governing factor in many of the Earth sciences. Relationships that are recognized at one scale may not hold at another scale, imposing a scale dependency recognized in many fields of science (Goodchild and Quattrochi, 1997). In geomorphology, spatially extensive approaches are needed to address global-scale concerns or to assess the implications of broad land-use changes. Schumm (1991) argued that the dimensions of time and space are linked. As the geographic extent of events increases, the average rates of effective change decrease and the time span to be considered increases; that is, as spatial scales of land-use change increase, more information is needed from historic (or prehistoric) reconstructions (**Fig. 1**):

> As the size and age of a landform increases, fewer of its properties can be explained by present conditions and more must be inferred about the past.
> (Schumm, 1991: 52).

Thus, the geomorphic dimension of global change must include historical perspectives of geomorphic science.

Geomorphology has a strong geologic-science component, and therefore, concepts of time and history have long been an essential part of understanding geomorphic features and processes (Thornes and Brunsden, 1977; Albritton, 1980). Depending on the purpose of an investigation, hillslope and river systems may need to be understood at a variety of timescales including Cenozoic (10^6–10^7 year), Holocene (10^3 year), historical (10^2–10^3 year), or steady time (10^0–10^1 year). The practices of geomorphology and engineering tend to part over the concept of time, which can be historic or evolutionary to geomorphologists but tend to be steady time to the engineer (James, 1999). In fact, geomorphologists recognize the potential for causality to shift on the basis of the timescale considered (Schumm and Lichty, 1965); that is, a dependent variable at short time scales (e.g., erosion depends on slope) may become an independent variable over geologic time (erosion determines slope). Geomorphologists have an arsenal of intellectual

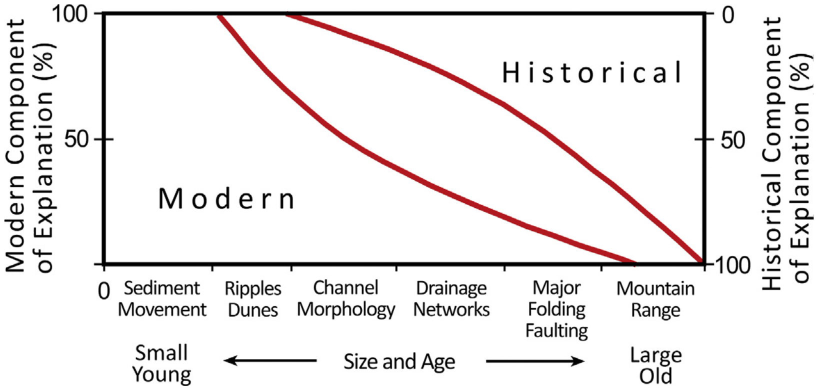

Fig. 1 Schematic diagram illustrating relevant timescales required to explain geomorphic processes at increasing spatial scales. To understand geomorphic systems at spatial scales relevant to global change studies, processes will need to be understood at historical timescales. Adapted with permission from Schumm SA (1991) *To Interpret the Earth: Ten Ways to be Wrong*. New York, NY: Cambridge University Press, 131 pp.

and analytical methods to deal with geologic time such as 'ergodic' reasoning (Paine, 1985), geomorphic equilibrium (Gilbert, 1877; Thorn and Welford, 1994), effective discharge (Wolman and Miller, 1960), and thresholds of stability (Schumm, 1979).

6.52.2.3 Effects of climate change on fluvial systems

Distinguishing the effects of past changes in land use from those of climate change can be difficult. Both may leave the same response (equifinality), both often occur simultaneously (polygenesis), and both leave incomplete evidence in the stratigraphic record. To the extent that a landscape may be geomorphically sensitive to land-use changes, it may also be sensitive to climate change. Moreover, climate change can increase landscape sensitivity to LU/LC change (and vice versa) and can result in land-use changes through such processes as migrations, adoption of new land-use practices, fire and flood frequencies, and so forth.

Earth-surface modelers have been fairly successful at modeling global-change processes through simulations of atmospheric and oceanic processes, but less successful with simulations of terrestrial systems that include lateral redistributions of water and sediment at Earth's surface (Pelletier et al., 2015). Complexities and uncertainties arise from the fact that Earth surface systems are highly non-linear, and that relationships between precipitation, vegetation, sediment production, and sediment transport capacity are poorly understood. One can add to that the fact that climate variability and the frequency of extreme events is more important to geomorphic change than are mean climatic values (Katz and Brown, 1992). The translation of abrupt climate change to geomorphic response involves the combination of several systems, which results in a non-linear process response. In addition, interactions between climate change and LU/LC change complicate interpretations of the record of past geomorphic processes and forecasts of future geomorphic change. Thus, applications of climate-change forecasts or hindcasts to geomorphic response is associated with great uncertainty. Potential impacts of climate change on LU/LC are discussed further in **Section 6.52.3.5**.

6.52.3 Hydrogeomorphic changes caused by land use

Upland land use affects watershed processes that govern the production, transport, and storage of water, sediment, nutrients, metals, and other pollutants. Human activities that disrupt vegetation and destabilize soils have the potential to decrease soil infiltration, suppress groundwater recharge, and amplify runoff generation and flood magnitudes. Altered flow pathways often increase flood frequencies, erosion, sediment production, and sedimentation, while degrading water quality and aquatic ecology. Increases in runoff produced by land clearance are proportionally greatest in small watersheds and can be subtle in watersheds larger than 5 km^2 in area (Potter, 1991; Jones and Grant, 1996; Jacobson et al., 2001). Hydrologic processes, such as infiltration capacities and hillslope runoff pathways, can be changed over a wide range of spatial and temporal timescales and vary between the various geomorphic systems that are covered in other chapters of this treatise.

6.52.3.1 Changes to flood regimes

Changes to runoff caused by land use generally generate changes in streamflow downstream that can be measured with hydrographs. The area under a hydrograph represents the volume or yield of water. The shape of storm hydrographs may change with hydrologic responses to land use. For a given yield, the shape can vary from a system characterized by gradually varying flows to

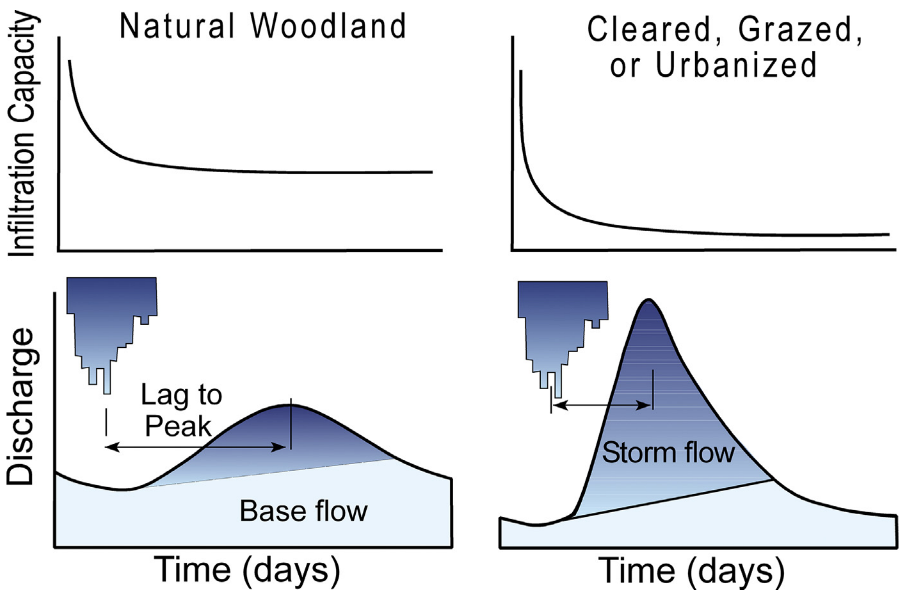

Fig. 2 Idealized infiltration curves and storm hydrographs for two contrasting small watersheds. Natural woodland (left) has high infiltration rates and generates hydrographs with low peak discharge, long lag-to-peak, and high base flows. An urban watershed (right) has low infiltration rates and generates Hortonian flows that cause flashy hydrographs with high peak discharges, short lag-to-peak, and low base flows.

a flashy system with high peaks and low base flows. Increased runoff generation contributes to larger storm flows and flood peaks downstream. As infiltration decreases with land clearance, surface runoff increases and hydrologic response becomes more variable with higher peak flows, shorter lag times, and lower base flows (Fig. 2).

Many studies have documented increases in water yield with deforestation (but not all, see Burt et al., 2015) using controlled experiments in paired watershed studies (Bosch and Hewlett, 1982; Brown et al., 2005; Hung et al., 2018a). Less is known about relationships between deforestation and extreme floods. Such an analysis should stratify events by magnitudes of storms and floods, watersheds by size, and land-use change by type and extent. During extreme rainfall intensities and durations when the infiltration capacity of forest soils is exceeded, saturated forested areas ultimately should produce as much runoff as their deforested counterparts, but such storm intensities may not be common. In general, high infiltration capacities and reduced runoff generation from forests during storms reduces flooding (Rogger et al., 2017). In a study of 56 developing nations, Bradshaw et al. (2007) found a positive correlation between flood frequencies and area of forest removal and a negative correlation between flood frequencies and area of forest remaining. Other studies have found that the effects of land-use change on floods are most pronounced in small watersheds with moderate-magnitude events (Tollan, 2002; Eisenbies et al., 2007). With regard to agricultural land uses, some crop types generate substantially more runoff than others. Examination of more than 45 years of runoff from a small watershed in the southeastern USA under changing land use indicates that row crops generated more runoff and higher peak discharges than kudzu, rescue grass, or Bermuda grass (Endale et al., 2006).

Responding to the concern that flooding is becoming more frequent and severe, Hall et al. (2014) reviewed the current knowledge on flood-regime changes in European rivers obtained through monitoring changes in the measured flood record and modeling of future flood scenarios. They suggest that a synthesis of these two approaches is needed to make progress in flood-change research. Although climate change has received considerable attention in terms of its effect on the increased frequency of major floods, fewer recent studies have examined the role of land-use change alone (Rogger et al., 2017). The effects of land-use change on floods involve complex interactions between processes at different spatial and temporal scales that make analysis and prediction difficult (Fig. 3). Plot-scale experimental studies suggest that compared to grassland, forest cover may reduce average discharge and lower flood peaks, whereas the effects of forest cover on flood peaks at the basin scale are less well understood (Rogger et al., 2017).

The timing and magnitude of flood peaks are not simply affected by local hillslope processes such as decreased infiltration and increased runoff. As the size of watersheds increases, floodwater storage and conveyance increase in importance. The factors that influence flood timing and magnitude at the watershed scale include storage in ponds, lakes, reservoirs, wetlands, floodplains, and broad valley bottoms. In large rivers, hydrograph shapes are determined by the arrival times of individual flood peaks from various tributary sources, each of which is influenced by storage elements. At this scale, processes can be greatly influenced by valley-bottom land-use practices that may alter floodwater storage. For example, wetland drainage, channelization, and levee construction decrease flood storage, accelerate down-valley conveyance, and shift downstream hydrographs toward a flashier response with high flood peaks (Opperman et al., 2009). Conversely, construction of dams or other storage reservoirs increases storage, decreases down-valley conveyance, and attenuates hydrographs downstream. In spite of these complexities, increased flood volumes from hillslopes and small watersheds generally produce higher flood peaks downstream. As larger floods are generated, the frequency of occurrence of a given size flood increases and its recurrence interval decreases. Thus, intensified land use in forested

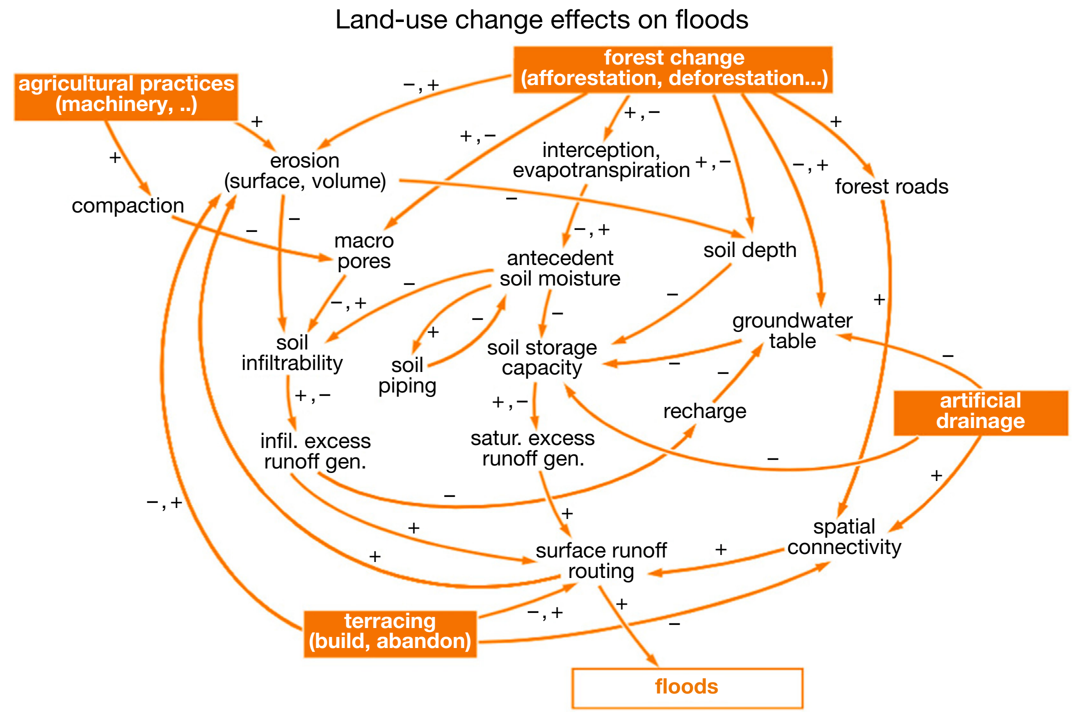

Fig. 3 Effects of land-use change on flooding. Schematic showing interactions between processes affected by land-use change at the catchment scale. Plus and minus signs indicate increasing or decreasing directions of change to the receiving variable. From Rogger M, Agnoletti M, Alaoui A et al. (2017) Land-use change impacts on floods at the catchment scale: Challenges and opportunities for future research. *Water Resources Research* 53: 5209–5219.

areas tends to involve deforestation that increases both the magnitude and frequency of flooding. Conversely, decreased intensity of agricultural land use is often associated with reforestation, the recovery of forests by planting (afforestation), or natural succession, which may reverse the changes in flood magnitudes and frequencies. The magnitude of these changes, however, depends on both the size of the watershed and the magnitude of the flood; large watersheds and large floods tend to be less affected (Benson, 1964; Knox, 1977; Pitlick, 1997; Wohl, 2000; Lecce and Kotecki, 2008).

The impoundment of streams by dams has a considerable effect on the flow regimes of large rivers through the storage and gradual release of peak flows (Williams and Wolman, 1984; Graf, 1999). This topic is covered elsewhere in this volume. The hydrologic effect of numerous small impoundments on headwater streams is far less well documented. The large number of farm ponds and other impoundments in the USA (Smith et al., 2002) plays a substantial role in reducing flood discharges when flood storage is increased in many small tributaries. The influence of small impoundments on sediment and nutrient deliveries in small watersheds is discussed at the end of **Section 6.52.3.3**.

The geomorphic effects of increased flood magnitudes can be substantial and enduring, although this varies greatly with scale, topography, types of channels, and duration of flooding. Magnitudes of peak discharge are not the only factor. Costa and O'Connor (1995) concluded that short-duration floods of a given stream power were much less geomorphically effective than floods of long duration, although short floods with extremely high peak flows can have very large effects (Magilligan et al., 2015). In mountain rivers, extreme floods can have devastating effects to geomorphic and aquatic ecological systems that are governed not only by stream power but also by valley-bottom configuration (Stoffel et al., 2016; Surian et al., 2016). Similarly, Thompson and Croke (2013) found that flood power for an extreme flood (~2000-year recurrence interval) in southeast Queensland, Australia, was 2–3 times lower in an unconfined valley reach (net depositional) than in a confined reach (net erosional).

6.52.3.2 Soil erosion

In addition to increased flood magnitudes downstream, reduced infiltration capacities may cause a shift toward Hortonian flows that can be highly erosive on uplands. The same land-use changes that decrease infiltration rates and magnify runoff leave soils vulnerable to severe erosion. Vegetation removal, agriculture, overgrazing, mining, road building, and construction tend to accelerate erosion by sheet flow, rills, and gullies (Poesen, 2018). Rills and gullies not only represent volumes of eroded sediment but also increase drainage densities by extending channel networks that concentrate flows and deliver water and sediment more efficiently downstream. Thus, changes in LU/LC may initiate severe episodes of soil erosion on uplands and sedimentation downstream. Historically, extensive land-use changes associated with colonization and deforestation have generated periods of accelerated erosion, sediment deliveries, and channel filling, followed by reforestation, decreased upland erosion, reduced sediment deliveries, and channel incision. This sequence can be described as an aggradation–degradation episode (ADE) (James, 2010; James and Lecce, 2013). The nature of ADEs and the legacy sediment that they may leave behind are described in **Section 6.52.4.2**.

Erosion by water results where and when forces applied by the fluid exceed the forces resisting erosion. The dominant applied fluvial forces are exerted by raindrop impacts and flows in sheets or channels. In modern terminology, especially in agricultural settings, raindrop splash and sheet erosion are often referred to as 'interrill erosion' (Nearing et al., 1989). Rill and interrill erosion have received the most research attention (Poesen, 2018). Although classic hydrologic theory emphasized sheetflow as the primary process in soil erosion (Horton, 1933), the importance of raindrop splash (Fernández-Raga et al., 2017) and the kinetic energy of precipitation was recognized later (Wischmeier and Smith, 1965, 1978). Raindrop splash dominates on interrill areas where it causes soil detachment and transport in all directions but with net movement downslope (Muchler and Young, 1975; Meyer, 1981; Ma et al., 2015; Zhang, 2019). Raindrop splash and sheetwash erosion may work together, but their relative importance shifts with increasing depths of overland flow. The translation of raindrop energy to soil particles is greatest in the absence of sheet flows, although transport of dislodged particles is facilitated by sheet flows. As the depth of sheet flow increases, erosion induced by raindrop splash may increase as dislodged particles are removed by sheet flow. As sheet-flow depths increase further, however, less energy from the raindrops reaches the soil surface and erosion by sheet flow begins to dominate. Raindrop splash is also minimized at a critical ponding depth (Gao et al., 2003). The effectiveness of raindrop splash and sheet flow to entrain and transport particles depends on the raindrop size, depth and velocity of sheet flows, vegetation cover, slope, grain size, grain cohesion, and organic matter. Land-use changes may substantially decrease resistance to erosion by removing vegetation, compromising root mats, and reducing organic matter. They may increase the applied forces by increasing runoff generation and concentrating flows into channels.

Several soil-erosion models have been developed to predict erosion or design mitigation strategies. Many models predicting soil erosion from land use are based on parameters of the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1965, 1978), such as the RUSLE model (USDA, 2013). This approach indicates that conversion of LU/LC from natural woodland to bare soil could result in three orders-of-magnitude increase in erosion for some soils (**Fig. 4**). Most of the sediment produced from such accelerated erosion is likely to remain near the site but it may reside in unstable deposits on over-steepened slopes from where it may be remobilized and conveyed farther downslope by subsequent erosional events. Alternatively, process-based soil-erosion simulation models such as WEPP (Flanagan et al., 2012) or EUROSEM (Morgan et al., 1998) are available that may be spatially distributed and modular in design (cf. Jetten et al., 1999) and may be used to estimate sediment production and yield at the catchment scale (de Vente et al., 2013).

6.52.3.3 Sediment yields and delivery ratios

River restoration and management programs tend to emphasize the effects of modified flow regimes on channel morphology and stability and neglect the importance of sediment regimes; i.e., the magnitude and frequency of sediment loadings over time (Wohl et al., 2015b). Human-altered sediment regimes are ubiquitous and knowing previous or future natural sediment regimes is difficult if not impossible, so river and environmental management should plan for sediment loads affected by human land use and other alterations. For geomorphic stability without long-term aggradation or degradation, a balanced sediment regime with yields close to transport capacity should be the target. Yet, understanding this balance requires recognition of complex non-linear sediment dynamics over a range of time scales. This section examines relationships between the amount of sediment generated by changes

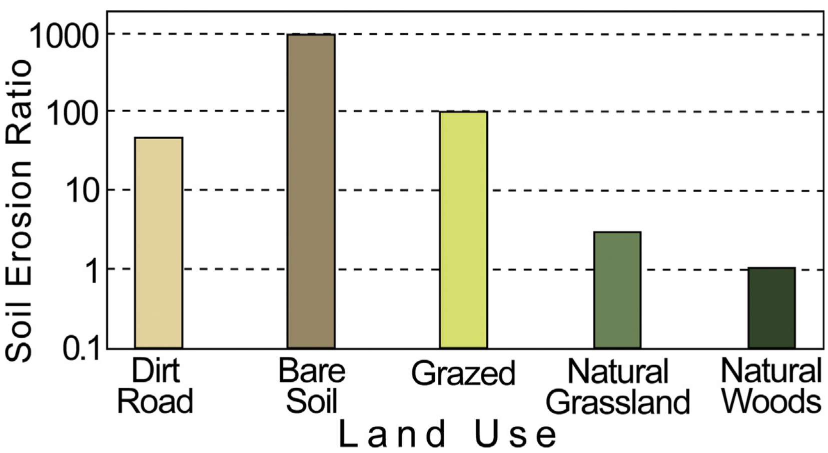

Fig. 4 Soil erosion can increase up to three orders of magnitude when land use is converted from natural woodland to more erodible land uses. Values are based on changing parameters in the Universal Soil Loss Equation. Adapted from Jacobson RB, Femmer SR, McKenney RA (2001) *Land-use changes and the physical habitat of streams: A review with emphasis on studies within the US Geological Survey Federal-State Cooperative Program/ USGS Circular No. 1175*, with permission from USGS.

in land use and downstream sediment yields. Processes associated with the mechanics of suspended sediment and bedload transport are covered elsewhere in this treatise.

6.52.3.3.1 Sediment delivery ratios

The amount of sediment eroded from hillslope systems; i.e., *sediment production*, is usually different than the amount of sediment passing a downstream point on the stream; i.e., the *sediment yield* at that point. Typically over the short term, only a small proportion of sediment produced by upland soil erosion reaches large river systems downstream (Walling, 1983; Fryirs, 2013). Much of the eroded soil is stored nearby as colluvium or as alluvium in fans, floodplains, wetlands, channels, or reservoirs behind dams (Walling, 2006). The relationship between sediment yield and production can be expressed as the 'sediment delivery ratio' (SDR):

$$\mathrm{SDR} = \mathrm{S_Y}/\mathrm{S_P} \tag{1}$$

where S_Y is the sediment yield and S_P is sediment production, which may be computed from soil erosion estimates and excludes lowland erosion of channels and floodplains. As so defined, the SDR represents the proportion of eroded upland soil that reaches the outlet of a watershed.

In most studies SDRs are less than unity, indicating sediment storage, which depends on factors controlling sediment transport, accommodation space, and the time since sediment pulses were generated (Sims and Rutherfurd, 2017; James et al., 2019). The large proportion of sediment stored in small watersheds carries several possible implications:

1. Large sediment repositories may be available for reworking. The stored sediment may be in colluvial deposits, floodplains, channels, wetlands, lakes and ponds. Reworking depends on factors that are highly variable, idiosyncratic, and related to connectivity. (Connectivity is described in **Section 6.52.4.1**.)
2. Low SDRs may reflect suspended sediment measurement errors, in particular underestimates of yields based on low measurements at stream gauges on large rivers. This is problematic because a large proportion of sediment is carried by extreme floods that are difficult to measure.
3. Low SDRs may be a transient condition in response to local storage in response to intensified land use that generated a pulse of sediment. Where this has occurred, increasing SDRs might be anticipated after conservation practices, agricultural field abandonment, and reforestation reduce sediment production and remobilization of stored sediment sustains high yields.
4. Values of SDR < 1 reveal a shift of sediment from hillslopes to adjacent lowlands that reduces local relief and gradients and represents a disequilibrium between hillslopes and fluvial systems. Under geologically stable conditions over millennial time scales–without substantial tectonic, sea level, or climate change—SDR should approach unity (Lu et al., 2005).

The SDR, in conjunction with erosion estimates, can be used to compute sediment yields in ungauged basins by rearranging Eq. (1) (Roehl, 1962; Renfro, 1975). The SDR, however, is essentially a black-box model that must account for all of the sediment storage and conveyance variability throughout the watershed; but it is governed by multiple variables and nonlinear processes. It is well known, for example, that SDR is scale dependent and often decreases downstream with increasing drainage area (e.g., Vanoni, 1975; Novotny and Chesters, 1989; Dendy and Bolton, 1976; Milliman and Meade, 1983; Walling, 1983; Lu et al., 2005; Worrall et al., 2014). In the USA—based on sediment yield data from the twentieth century—SDRs varied from $> 50\%$ in small watersheds less than 1 km^2 in area to 10% or less in watersheds greater than 100 km^2 (**Fig. 5**). However, numerous studies have shown that that

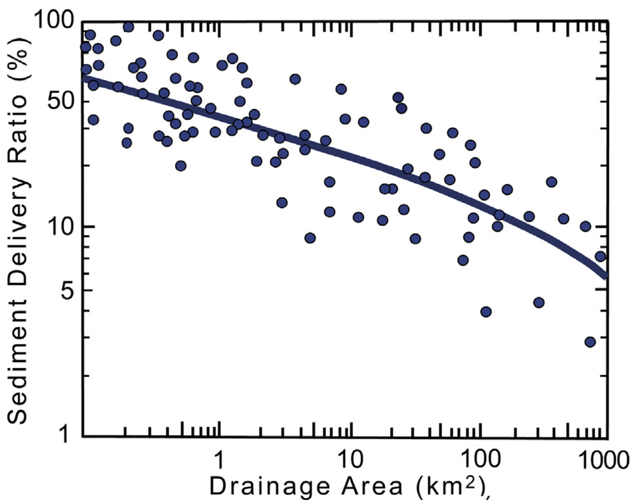

Fig. 5 Sediment delivery ratios as a function of drainage area. Most sediment produced in uplands does not leave small watersheds. Adapted from Novotny V and Chesters G (1989) Delivery of sediment and pollutants from nonpoint sources: A water quality perspective. *Journal Soil and Water Conservation* 44(6): 568–576; cf. Vanoni VA (ed.) (1975) *Sedimentation engineering. Manuals and Reports on Engineering Practice No. 54*, New York: ASCE.

the relation between drainage area and S_Y can be negative, positive, or both (see review by de Vente et al., 2007) and that considerable scatter occurs in plots of S_Y and drainage area, which indicates the importance of other factors, such as LU/LC. For example, Dedkov (2004) found a positive relation between drainage area and S_Y can be attributed to well-developed vegetation cover, limited human disturbance, and channel erosion as the dominant source of sediment, whereas a negative relation was found for intensely cultivated basins (de Vente et al., 2007).

A standard metric for suspended sediment yields that is adjusted for scale is the area-weighted *specific sediment yield* (SSY; t km^{-2} $year^{-1}$). Yet, many studies have shown that even SSY is scale dependent (Milliman and Syvitski, 1992; de Vente et al., 2007). Church (2017) noted that the effect of drainage area on SSY is often not recognized, so many comparisons of sediment yields are invalid. He proposed that a rational SDR could be constructed using upscaled sediment yield data. The dependency of SDR and SSY on spatial scale is highly variable when international studies are considered (de Vente et al., 2007). For example, Diodato and Grauso (2009) computed SDRs for 25 basins in Italy and 11 international basins and found no correlation with drainage area. The high variance in SDR may result from variations in climate, tectonics, soils, vegetation, or land use (James, 2018). Charts such as **Fig. 5** can provide guidance for interpreting the values of SDR, although this is an imprecise practice that provides, at best, a first approximation (Parsons et al., 2006).

LULC conditions and human activities often determine whether hillslope erosion or channel erosion is the dominant source of sediment (de Vente et al., 2007). Where hillslope sediment sources are dominant, SDRs are likely to decrease with drainage area, but where channel erosion is dominant, SDRs may increase with increasing drainage area. Shi et al. (2014) found that variables expressing land-use composition and patterns were important determinants of SDR in the 8973 km^{-2} upper Du River watershed of China. Time scale is also a factor that drives SDR values. Although SDR was originally a short-term, interannual concept (Parsons et al., 2006), it is often applied to centennial or longer periods, so they should be scaled to time (Lu et al., 2005). Moreover, SDR values have been shown to decrease through time following major sedimentation events (James et al., 2019). Low SDRs maintained over geologic time would have implications to geomorphic theory. Storage of most sediment production in nearby small watersheds indicates a state of disequilibrium between hillslopes and the fluvial systems to which they are coupled (Trimble, 1977; Walling, 1983). An equilibrium was commonly assumed in early long-term denudation studies that assumed present sediment yields could be used to hindcast rates over geologic time (Judson, 1968). If SDRs had been continuously low over geologic time, however, upland erosion and valley filling ultimately would have reduced local slopes and relief to zero. Theoretically, low SDRs are not sustainable over geologic time if hills and valleys are to be maintained.

Traditional views of the SDR have received criticism in recent years as being flawed for a number of reasons (Parsons et al., 2006; de Vente et al., 2007; Worrall et al., 2014; Hoffmann, 2015; Church, 2017). Parsons et al. (2006) argue that although the SDR is useful for estimating sediment storage at the spatial scale of drainage basins and the temporal scale of a year, it is not conceptually sound because it is not supported by observations at larger and smaller temporal and spatial scales. They argue further that the decline of the SDR with drainage area is not because of increased storage in larger basins, but that it simply takes longer for sediment to travel greater distances. Similarly, Worrall et al. (2014) suggest that the negative relation between the SDR and drainage area is not necessarily caused by downstream reductions in upland sediment deliveries, but may represent a lack of independence between SSY and drainage area (self-correlation) or changes in the production of sediment from channels.

6.52.3.3.2 Importance of storage to long-term sediment budgets

Sediment budgets and computations of sediment yields in ungauged basins require estimates of sediment storage that vary greatly through space and time. Sediment storage potential is not well understood, so this is an area where research is needed. The importance of geologic controls on accommodation space (e.g., valley width) is one important factor (Magilligan, 1985; Lecce, 1997; Brierley, 2010). The importance of local sediment storage introduces a great deal of uncertainty to sediment yield estimates and sediment storage potential on valley bottoms firmly weds watershed hydrology to fluvial geomorphology. Recognizing spatial variations in storage and differentiating temporary storage sites from sediment sinks are essential to river management (Fryirs and Brierley, 2001).

An implication arising from the common dominance of sediment storage in small watersheds is that anthropogenic sediment production has been much greater than previously estimated. A central thesis of this chapter is that the impacts of land use on fluvial systems during colonization and industrialization involved major increases in runoff and sediment production. Understanding these impacts over a variety of spatial and temporal scales requires a sophisticated appreciation for sediment storage. Early recognition of the importance of human impacts on sediment production was based largely on suspended sediment data (Douglas, 1967; Meade, 1969; Milliman and Meade, 1983). These computations tended to compensate for large dams, but they did not adequately compensate for storage in small watersheds (Trimble, 1977). Thus, initial estimates of human-induced sediment production may have been grossly underestimated.

Local storage of anthropogenically accelerated sediment production may explain much of the disequilibrium indicated by low SDRs and supports the implication that widespread low SDRs are a geologically recent and transient condition. Land-use changes, generally in conjunction with late-Holocene climate change, introduced vast amounts of sediment that have not yet worked through fluvial systems (e.g., Hoffmann, 2015). Where land-use change and local storage cause low SDRs, a shift toward increasing SDRs might be anticipated as conservation practices and reforestation reduce sediment production and channels increasingly exploit reservoirs of stored sediment on valley bottoms. Fryirs and Brierley (2001) described a case in the Bega catchment, New South Wales, Australia, where SDRs to the lowlands are relatively high (almost 70%), yet most of the sediment remains stored there and has not made it through the estuary to the ocean. The importance of local storage of large sediment pulses suggests that forecasts and management of sedimentation rates, reservoir infilling, stream habitats, water quality, and other sediment-related phenomena should anticipate spatially and temporally complex processes that govern the gradual release and downstream arrival of sediment (Sims and Rutherfurd, 2017).

6.52.3.3.3 Calculating and modeling sediment yields and budgets

Many models predict soil erosion and sediment yield (see reviews in de Vente and Poesen, 2005; de Vente et al., 2013; Vigiak et al., 2015; Pandey et al., 2016). Spatially distributed simulations of land-use effects on erosion and sediment deliveries in moderately large watersheds (500–2000 km^2) can be made with models that include sediment storage allowances and can be calibrated with large available data sets. For example, the Soil Water Assessment Tool (SWAT) model was developed by the US Department of Agriculture Agricultural Research Service for use with broad-based data sets. The SWAT model was developed to predict impacts of land management on water, sediment, fertilizer, and pesticide yields in large ungauged basins (Arnold et al., 1998). It can use State Soil Geographic Database (STATSGO) (1:250000 scale) or Soil Survey Geographic Database (SSURGO) (1:250,000 scale) soil data from the USDA Natural Resources Conservation Service (NRCS) with land-use data from the USDA National Agricultural Statistics Service (NASS) or from the US Geological Survey (USGS) National Gap Analysis Program (GAP). It uses the Soil Conservation Service (SCS) curve number method to estimate runoff from daily precipitation, and simulates infiltration, evapotranspiration, channel routing, and groundwater flows. The model is spatially distributed to the sub-basin scale (Leavesley et al., 1983), but lumped within sub-basins. At broad scales of study, the quality of land-use and soil data is not ideal, and much of the uncertainty associated with runoff modeling at this scale arises from estimates of input parameters derived from soil and land-use data (Heathman et al., 2009).

The time scale of concern has a great bearing on estimates of sediment yields. Methods intended for annual fluxes may not be appropriate for analysis of longer periods and vice versa, so sedimentologic evidence of long-term sedimentation events is of greater importance. Over long periods of time, uncertainties are introduced by dating techniques, but with the proper use of isotope concentrations and other methods, accurate histories of sediment production and deposition can be constructed. For example, relative and absolute dating of sedimentary deposits can be used to constrain past rates of soil erosion. Over the long term (10^3–10^5 year), methods include cosmogenic nuclides such as ^{10}Be and ^{26}Al (McKean et al., 1993; Small et al., 1999; Heimsath et al., 1999, 2005), optically stimulated luminescence (Wallinga, 2002), and traditional dating tools such as ^{14}C and correlations with cultural artifacts. For erosion and sedimentation rates over historical time periods, fallout radionuclides can be used (Quine and Walling, 1993; He and Walling, 2003). The use of ^{137}Cs isotope concentrations in erosion studies can identify stable surfaces as well as distinguish and partition surface soil from subsoil as sediment sources in young deposits (Ritchie et al., 1974; Walling and Woodward, 1992; Wallbrink and Murray, 1993). This method may allow historical reconstructions of the onset and cessation of gullying in a watershed during the historic period (Olley and Wasson, 2003). Radionuclides may also be used to identify sediment sources. Wilson et al. (2008) used a mixing model based on ^{7}Be and ^{210}Pb radionuclides in small watersheds to determine the relative contributions of upland versus channel sources in suspended sediment. They found that sediment from soil erosion was prevalent early in runoff events, but, later in storm events, channel contributions became dominant. Hillslope sediment production can be quantified under various land uses, along with rates of storage in colluvial deposits along valley margins, in alluvial deposits on floodplains, and within channels (Vandenberghe and Vanacker, 2008).

Dams may have a tremendous effect on downstream sediment deliveries in fluvial systems by storing sediment and, in some cases, negating the effects of accelerated anthropogenic erosion. Many global-scale studies of sediment yields are based on deliveries to the oceans or to coastal zones and, therefore, include large reductions in yield caused by storage in reservoirs (Syvitski, 2003; Syvitski et al., 2005; Walling, 2006). This may result in sediment yields prior to human disturbance being greater than modern yields near coasts (Syvitski et al., 2005), in spite of substantially higher modern upland production. From an upland land-use perspective, the proliferation of small ponds can also reduce sediment and nutrient deliveries (Smith et al., 2002; Renwick et al., 2006; Verstraeten and Prosser, 2008). Whereas previous studies concluded that colluvial and alluvial deposits are the primary sinks for sediment eroded from uplands, small farm ponds can be a major sediment sink. For example, Renwick et al. (2005) showed that small impoundments in the USA are important components of sediment budgets, capturing an estimated 25% of sediment produced by sheet and rill erosion.

6.52.3.4 Impacts of urbanization

Urbanization is a particularly intensive land-use change resulting in a suite of environmental alterations associated with industrial, commercial, and residential developments on previously undeveloped or agricultural lands. Urbanizing watersheds involve increasing population densities, built areas, infrastructure, and hydrologic alterations (Alberti, 2005; Booth and Bledsoe, 2009; Kaushal and Belt, 2012), which all contribute to geomorphic changes and contributions of contaminants. Historically, studies of urban impacts on hydrologic and geomorphic systems have focused on development concentrated in urban or suburban centers. Land-use change often involves geographically extensive application of urban development processes, such as construction, pavement, and storm sewers, in a diffuse manner that is not concentrated in urban centers. Understanding the processes and impacts of urbanization on geomorphic systems applies, therefore, not only to urban areas but also to more widespread land-use changes.

6.52.3.4.1 Hydrologic effects of urbanization

Hydrological alterations in urban streams may produce increases in flow depth, velocity, and sediment transport capacity far beyond pre-development conditions, resulting in bed scour, channel incision and enlargement, and reduction of channel complexity (Wolman, 1967; Chin, 2006; Russell et al., 2017, 2020). The "urban stream syndrome" has been used to describe the overall degraded conditions common to many urban streams, including ecological impacts as well as geomorphic changes (Walsh et al., 2005). Although urban disturbances and channel impacts are often similar between systems, specific geomorphic responses to urban land use can vary because of differences in hydrologic and geologic controls even among streams in the same watershed (Booth et al., 2016). Urbanization increases runoff rates and flood peaks by several times the predevelopment condition because of the combined effects of increased impervious cover, which dramatically reduces precipitation infiltration rates, and expansion of artificial drainage networks, which rapidly concentrates and deepens flows downstream in a watershed (Leopold, 1968; Li and Wang, 2009; Hung et al., 2018a). The style of urbanization can also determine magnitudes of runoff increase. For example, Brander et al. (2004) found that urban cluster development of residential subdivisions generated a lower volumetric increase in runoff than in curvilinear, coving, or new urbanism styles of suburban development owing to a greater area of natural permeable surfaces retained in neighborhood design.

A common measure of the degree of urbanization in a watershed is the "percent impervious area" (PIA) covered by roads, roofs, parking lots, sidewalks, and other built surfaces, which is generally related to population density (Stankowski, 1972; Reilly et al., 2004). In general, when PIA exceeds 10% of the watershed, negative effects on riparian ecology (including channel alterations) begin to appear, and at percentages greater than 30% the impacts become acute and widespread (Klein, 1979; Arnold and Gibbons, 1996; Goetz et al., 2003; Cianfrani et al., 2006). Impervious area is a sensitive predictor of urban hydrology in smaller watersheds with steep hillslopes that originally contained highly permeable soils and denser vegetation compared to larger watersheds (Bledsoe and Watson, 2001; Hung et al., 2018a; Ress et al., 2020). However, PIA is not always a good predictor of urban channel response (Vietz et al., 2014, 2016a,b; Russell et al., 2018; Taniguchi et al., 2018). A better indicator of urban hydrologic response combines the effects of drainage connectivity and impervious area by using percent "effective impervious area" (EIA), which quantifies the relative amount of impervious areas directly connected to storm drain systems in a watershed (USEPA, 2000). Compared to the more general total impervious area, EIA was found to be a better predictor of hydrologic disturbance in urban watersheds with the threshold of geomorphic response reported to be 2–3% (Vietz et al., 2014). Moreover, infrastructure connections to channels such as storm drains, concrete-lined channels, and road crossings that deliver runoff to the channel system can provide a framework for the identification of channel disturbances and selection of management options for urban streams (Gregory and Chin, 2002).

New stormwater management methods and technologies can greatly reduce the hydrologic impact of urban land use. These practices, generally known as 'low impact development' (LID), include the use of permeable paving materials and the diversion, detention, or retention of storm water to permeable areas. Permeable materials can reduce surface runoff to almost zero (Brattebo and Booth, 2003). Impervious areas may also be reduced by the use of vegetated (green) rooftops, landscaping with 'rain gardens,' and other on-site collection methods that detain stormwater and encourage infiltration. Conventional urban development conveys runoff away from the site, whereas LID projects integrate landscaping and green space to mimic natural infiltration processes onsite. By encouraging infiltration and groundwater recharge, LID can mitigate the harmful effects of urban runoff generation. LID methods may also cost less than conventional development by reducing the need for storm sewers and culverts. Simulating the effects of urban development on runoff with spatially distributed models may require special adjustments for diversions of runoff from impervious surfaces to rain gardens or detention structures (Moglen, 2000; Holman-Dodds et al., 2003). Although LID systems are effective for controlling flood waters from frequent, less-intense storms, local and regional detention systems may more effectively control larger floods with high peak flows (Giacomoni et al., 2014).

6.52.3.4.2 Geomorphic effects of urbanization on fluvial systems

Research on the effects of urban land use on channel form and stability and sediment yields in the USA began in the 1960s (Wolman, 1967; Leopold, 1968; Guy, 1970). Wolman's (1967) seminal work described how land-use changes through the transition of forested, agricultural, and urban phases in the Maryland and Pennsylvania Piedmont affected runoff, channel form, and sediment loads. Urban land use greatly increased sediment loads and rates of channel widening during and immediately following the construction period. Yields returned to moderate levels following urban landscaping, revegetation of abandoned farmland, and increase of channel widths enough to convey the additional runoff (Wolman and Schick, 1967). In an innovative study, Hammer (1972) used a detailed sampling approach and regression analysis to analyze the effects of land use and topographic characteristics on channel enlargement for 78 small urban and 28 rural watersheds in the Pennsylvania Piedmont. Using a "channel enlargement ratio," he found that channel cross-section areas had increased by >5 times in reaches with higher basin slope, greater proximity of impervious surfaces to the channel, and more frequent artificial channel improvements. Little channel enlargement occurred below developments younger than 4 years old, presumably indicating a lag time for incision and head-cutting to occur, or greater than 30 years old, suggesting that older developments had less efficient drainage systems and were covered with more vegetation (Hammer, 1972). In Denver, Colorado, USA, Graf (1975) used aerial photography and field data to describe the geomorphic response of ephemeral streams draining urbanizing watersheds, reporting an initial phase of aggradation and lateral expansion of inundation and deposition on floodplains followed by channel incision and transport of excess sediment downstream.

Fluvial geomorphologists provided important contributions to improve the understanding of channel response and sediment supply/transport in urban watersheds. Calls for more research on urban discharge and sediment problems to address environmental planning concerns were made in the early 1970s (Guy, 1970; Leopold, 1973). Almost a decade later, Graf et al. (1980) presented several new opportunities for research in fluvial geomorphology focused on human interactions with stream channels including urban systems. Almost 30 years later, a large multi-disciplinary panel published a list of key research questions to address knowledge gaps in urban stream ecology including several geomorphological topics: (i) hydrologic budgets, responses, and relationships to specific urban land-use and management practices; (ii) channel and sediment responses at different stages of development including potential for a new stable state; (iii) downstream effects of artificial infrastructure including piped and concrete-lined channels on flows and channels; and (iv) response of urban channels to climate change (Wenger et al., 2009).

Geomorphologists have contributed greatly to understanding the spatial and temporal variability of channel form and sediment responses to urbanization and imposed hydrologic alterations. In England, streams draining urbanizing areas had bank erosion rates 3.6 times greater and head-cut migration rates 2.4 times greater than rural streams (Neller, 1988). Channel enlargement was greatest in reaches with higher channel slope and more road crossings (Neller, 1989). Channel capacity enlargements can be spatially discontinuous and can involve widening, deepening, or a combination of the two (Gregory et al., 1992). In the Pacific Northwest, steeper channel slopes and weaker geologic substrates were associated with higher rates of channel incision and head cut migration in streams draining urban areas (Booth, 1990). In the same region, relatively stable streams typically drained more forested area, intact riparian buffers with forest or wetland, and reaches without road crossings (McBride and Booth, 2005). In Kentucky, channels in urbanizing watersheds became wider and deeper, pools became longer and deeper, riffles got shorter, and bed material became coarser in the early stages of development (Hawley et al., 2013). In southern California and nearby Mexico, the cross-section area of semiarid channels increased up to 64 times the pre-development condition in response to urban development (Trimble, 1997; Hawley and Bledsoe, 2013; Taniguchi et al., 2018). Natural restabilization of stream channels in urban watersheds can take 10–20 years, but recovery does not always follow trends in magnitude or expansion of urbanization (Henshaw and Booth, 2000; Booth and Fischenich, 2015). Local resistance factors including bedrock control, cohesive substrates, and riparian vegetation can mitigate the effects of urbanization on stream channels (Kang and Marston, 2006).

Local factors can exert significant control over the location of channel disturbances in urban streams. For example, in southern California the relative degree of channel enlargement increased upstream from channel hard points such as grade controls and bedrock outcrops (Hawley and Bledsoe, 2013). In Tijuana, Mexico, however, channel enlargement increased greatly downstream from channel hard points (Taniguchi et al., 2018). In general, channel morphology and bed sediment characteristics differ among forested, agricultural, and urban watersheds with similar geology and climate (Wolman, 1967; Shepherd et al., 2010; Laub et al., 2012). However, all channels in an urban watershed may not respond in a similar manner (i.e., head-cutting and enlargement). Indeed, studies comparing urban and rural channel conditions typically find some overlap in form and sediment properties because of similarities in local slope, substrate, vegetation, and management factors (Pizzuto et al., 2000; Bledsoe and Watson, 2001; Hession et al., 2003; Niezgoda and Johnson, 2005; Cianfrani et al., 2006; Galster et al., 2008). Relationships between land use and stream response can be complicated by: (i) covariation of anthropogenic and natural gradients; (ii) existence of multiple, scale-dependent processes; (iii) nonlinear responses; and (iv) influence of historical factors on present conditions (Allan, 2004).

Geomorphic models of stream-channel response in urban watersheds can be improved by including variables other than impervious area, such as the ratio of disturbing to resistive forces, proximity to geomorphic thresholds, and channel evolution trajectory (Bledsoe et al., 2012; Taniguchi et al., 2018). Indeed, channel evolution models for urban streams typically describe a single-channel response initiated by erosive runoff with subsequent incision, bank erosion, widening, and formation of a lower re-stabilized channel. However, other less-studied channel responses also occur in urban watersheds because of the influence of (i) higher sediment loads causing bed aggradation, (ii) vertical and lateral confinement on channel activity, (iii) multithreaded channel systems, and (iv) riparian vegetation controls (Booth and Fischenich, 2015).

6.52.3.4.3 Urbanization and sediment yields

The effect of urbanization on sediment inputs to stream channels has been a traditional subject of inquiry by geomorphologists (Wolman, 1967; Wolman and Schick, 1967; Graf, 1975; Trimble, 1997). Sediment transport capacity can be up to three orders of magnitude higher in urban streams compared to those draining forests (Russell et al., 2020). Typically, bank and bed erosion is the major source of sediment in urbanizing streams (Trimble, 1997; Gellis et al., 2017; Taniguchi et al., 2018). More than half of the annual sediment load to an urban stream can come from upland drainages, construction sites, and channels (Wolman and Schick, 1967; Trimble, 1997; Gellis et al., 2017; Russell et al., 2019a; Malhotra et al., 2020). In an urban watershed (31 km^2) on the southern Piedmont in Alabama, sand from bank erosion was the major source of the sediment load in the upper segments, whereas silt and clay from construction sites was the major source to the lower segment (Malhotra et al., 2020).

The prevailing model of the timing of sediment responses to urbanization holds that suspended sediment loads are extremely high during the construction period, but decrease to near background levels within a decade or two (Wolman, 1967). According to this model, supply rates decay with increasing impervious surface cover, maturing landscaping, and channel adjustments to a new quasi-stable state. A review of urban sediment-load data from 48 studies with 334 data points generally validated the Wolman (1967) model, but found that sediment yields from established urban areas tended to remain high and above background levels because of erosion of available sediment from artificial and local substrates and increased transport by stormwater runoff from impervious surfaces (Russell et al., 2017).

Recent studies in urban and suburban watersheds in Melbourne, Australia, provide new insights into the sources and transport of bed sediment. Construction sites produced 32% of coarse (>0.5 mm) sediment load from only 0.5% of the urban watershed area. However, the ratios of the sediment yield from different urban surfaces to the yield from impervious areas (2.1 Mg km^{-2} yr^{-1}) increased in the order: impervious area (1), grass-mulch area (4), gravel surfaces (35), and infilled construction sites (133) (Russell et al., 2019b). Bedload yield and particle size increased with intensity of urbanization and more specifically with effective imperviousness and drain-pipe density (Russell et al., 2018). A substantial source of bedload to the stream was from imported fill materials and artificial surfaces. Interestingly, there was no evidence that urban land cover severely limited sediment supply. A coarse sediment budget for a suburban watershed (5.5 km^2) showed that 36% of the sediment delivered to the stormwater network was exported downstream out of the watershed with the rest being deposited in the sediment cascade and either stored or removed from the catchment by sedimentation ponds (Russell et al., 2019a).

Only a few studies have been conducted of large wood recruitment, load, and transport in urban streams. Workers in northeastern Ohio, USA, observed that increasing PIA resulted in smaller wood size, lower wood abundance, fewer jams, and increased wood mobility of up to hundreds of meters in urban streams (Blauch and Jefferson, 2019). Further, the supply of wood in urban streams was lowest in confined and modified stream corridors, with narrow forested buffers generally unable to maintain stable wood structures in the channel (Blauch and Jefferson, 2019). The reduction of wood loads in urban streams compared to rural streams is partially related to removal by managers as well as lower recruitment rates. Efforts to increase the density of large woody debris (LWD) in urban streams can improve geomorphic stability of the channel over periods of 2–10 years (Larson et al., 2001). Instream log placement in urban streams in the Pacific Northwest increased the number of stable wood structures, lowered pool number, shortened pool spacing, and had little effect on controlling sedimentation (Larson et al., 2001). Moreover, to reduce the need for wood removal programs in California, the practice of modifying culverts and bridges to allow wood passage during floods was assessed based on economic and engineering feasibility and geomorphic effects (Lassettre and Kondolf, 2012). Overall, passage modifications were similar in cost to wood removal in the short-term (1–50 years) but became less expensive in the long-term (51–100 years) as the lower need for infrastructure replacement reduced flood costs and habitat loss (Lassettre and Kondolf, 2012).

6.52.3.4.4 Remote assessment and management of urban streams

Efforts to standardize methods to improve the precision and accuracy of geomorphic channel assessments in urban watersheds increased after 2000 (Montgomery and MacDonald, 2002). Although there were a few previous contributions (e.g., Gregory et al., 1992), advanced field and remotely-supported protocols were developed to evaluate various channel form and process properties including channel stability (Doyle et al., 2000), erodible river corridor widths (Piegay et al., 2005), physical heterogeneity (Reid et al., 2008), and susceptibility to hydromodification (Bledsoe et al., 2012). Further, hydrologic and geomorphic studies were expanded in scope to cover multiple sites across regions (Poff et al., 2006; Fitzpatrick and Peppler, 2010; O'Driscoll et al., 2010). Even though aerial photographs were used early on (e.g., Hammer, 1972; Graf, 1975), remote-sensing methods, such as digital processing of multispectral imagery, improved the accuracy of land-use classifications and mapping of impervious surfaces (Fankhauser, 1999). Several types of remote-sensing imagery have been utilized for this purpose, and methods were improved by combining multi-sensor or multi-image analyses (Carlson and Arthur, 2000; Wu and Murray, 2003). For example, the combination of aerial photography and Light Detection and Ranging (LiDAR) data greatly improved impervious mapping, especially with the addition of vegetation cover heights derived from LiDAR (Hodgson et al., 2003; Hung et al., 2018b). In addition to directly mapping impermeable surfaces such as rooftops and roads, surrogate parameters that influence impervious surfaces in residential areas can be measured, such as lot size, residential capacity, street width, and intersection density (Stone, 2004). Structure-from-Motion photogrammetry is also becoming more common for use in urban channel assessments (Taniguchi et al., 2018).

Attempts to reduce urban impacts by treating the channel (i.e., channel reconfiguration) to convey and resist the imposed runoff are not typically effective in reducing channel network instability, improving water quality, or restoring channel habitats and biota (Vietz et al., 2016b). Further, a restoration framework that relies only on generalized geomorphic assumptions and a "one size fits all" approach to hydrological assessments, stormwater improvements, and channel management is not sufficient (Laub et al., 2012; Giacomoni et al., 2014; Booth et al., 2016; Vietz et al., 2016a). The causes of excessive urban runoff and alteration of the sediment regime should be addressed using watershed-wide practices and not by treating the symptoms with armored channels (Vietz and Hawley, 2019). Channel controls based on geomorphic principles such as channel widening, grade control, increasing roughness, and sediment augmentation might be effective in reducing the negative effects of urban runoff but can be difficult to design, install, and maintain (Russell et al., 2020).

Urban channel management and restoration is challenging. In Maryland, streams draining urban catchments had channel complexity metric scores that were higher or equal to nearby forested catchments and generally exceeded scores from restored stream channels (Laub et al., 2012). In the southeast Piedmont of North Carolina, the condition of restored streams was not any better than nearby degraded urban streams, with forested streams in the best condition (Violin et al., 2011). In the southern Appalachian Mountains, suburban stream conditions were similar to surrounding agricultural streams prior to development (Burcher and Benfield, 2006). In general, geomorphic conditions of streams in urbanizing watersheds, particularly those of ecological importance, can only be maintained if excess stormwater flows are kept out of streams through retention, infiltration, or harvesting (Hogan et al., 2014; Vietz et al., 2014).

6.52.3.4.5 Urbanization and contaminants

Urban watersheds can have unique channel, sediment, and contaminant characteristics (Douglas, 1985; Taylor and Owens, 2009; Booth and Fischenich, 2015; Dijkstra et al., 2019). Large-scale assessments of urbanized basins have stressed the role of human activities in influencing sediment geochemistry and chemical budgets. A comprehensive study including 51 river basins, 1200 bed sediment samples, and analyses of a variety of major and trace elements in the <63 um fraction, reported that bedrock type had a limited influence on background geochemistry. Only the percent urban land and population density in the watershed were related statistically to trace metal concentrations in river sediments (Horowitz and Stephens, 2008). Moreover, sediment metal concentrations in urban-industrial centers in the Seine River Basin were three orders of magnitude greater than those in relatively undisturbed headwater drainages (Horowitz et al., 1999). Each metal exhibited its own specific source, pathway, and storage modes (Thevenot et al., 2007). However, metal budgets for the Seine region indicated that (i) waste-water discharges constitute a small portion of the total metal burden released to the basin, (ii) storage of particulate metals in reservoir sediments, floodplains, and dredged spoils were relatively minor compared to storage in soils, landfills, and structures, and (iii) the control of stormwater runoff is important to reduce metal concentrations in the river (Thevenot et al., 2007). Landfill leachates and cover erosion can provide a substantial source of metal pollution to local streams (Mantei and Coonrod, 1989), and high concentrations of metals can accumulate in bed sediments in streams below urban developments less than 30 years old (Thoms and Thiel, 1995).

Three types of pollution sources are generally found in urban areas: (i) nonpoint contaminants deposited on impervious surfaces and accumulated as salts, organic residues, and particles until released to stream channels during runoff events with little opportunity for natural filtering; (ii) point-source inputs of wastes released directly by commercial, industrial, and water-treatment facilities; and (iii) contaminated sediment from erosion of contaminated soils, particles from the weathering of artificial surfaces, and remobilization of previously contaminated fluvial deposits (Rhoads and Cahill, 1999; Sutherland, 2000; Martin, 2004; Bain et al., 2012; Pavlowsky, 2013). Particulate metal deposition in urban streams tends to occur within the active channel and along adjacent channel banks reflecting the limited extent of deposition of pollutants from point source discharges during low flow periods and first-flush sediment pulses (Martin, 2004; Taylor and Owens, 2009; Matys Grygar et al., 2013). Further, artificial fill materials dumped along and within valley floors for development purposes can be subjected to erosion by channel and overbank flows and thereby release contaminated sediment to urban streams (Bain et al., 2012). Reduction of floodplain area available for overbank deposition through confinement by artificial fill and structures along urban streams can increase in-channel contamination levels and downstream transport rates.

Road-deposited sediment (RDS) provides a unique source of contaminated particles to urban streams. RDS is created by the weathering and breakdown of artificial surfaces composed of material like asphalt, cement, and mixed fill materials (Taylor and Owens, 2009). In a highly urbanized area within the Aire-Calder River watershed in West Yorkshire, England, Carter et al. (2003) used sediment fingerprinting to identify the distribution of five fluvial sediment sources: channel bank erosion (18–33%), uncultivated top soil (4–7%), cultivated topsoil (20–45%), road deposited sediment (19–22%), and sewage input (14–18%). Calcium from the erosion and dissolution of cement and other pavement sources was found to be enriched in urban stream sediments, compared to rural streams, in Baltimore, Maryland (Bain et al., 2012). Sand-sized particles in road sediment may contain more residual phase metals that are less available to biota, but finer particles or organic matter can bind metals in higher concentrations in more available phases (Dong et al., 1984; Sutherland, 2000; Sutherland et al., 2000).

Polycyclic aromatic hydrocarbons (PAH) are a particular problem in some urban areas. Surfaces coated by coal-tar sealants can break down in response to traffic and runoff abrasion, which increases total PAH concentrations in sediments to levels of concern in nearby streams and drainage systems (Mahler et al., 2005). Parking lots covered with coal-tar sealants supplied at least 80% of the total PAH concentration in urban stream and pond sediments in the Galloway Creek watershed in southwest Missouri (Pavlowsky, 2013). Sediments from below coal-tar treated lots had PAH_{16} concentrations 35 and 480 times greater than those of unsealed asphalt and concrete lots, respectively. Using a drainage area-scaled, dilution-based regression approach, sediment PAH concentrations in multiple sub-watersheds were found to be highly correlated with the percentage of sealed parking lot area within the upstream drainage area of the sampling site, but poorly related to total parking lot area, sediment size, and sorting influences. In south-central Pennsylvania, sediment PAH concentrations were three times higher in urban compared to rural streams and highly correlated with combined residential/commercial/industrial land use (Witter et al., 2014). Further, source signature tracing indicated a strong relationship between coal-tar sealcoat dust composition and the distribution of sediment PAH concentrations in urban streams (Witter et al., 2014). Although reservoir sediments have yielded important PAH pollution records (Van Metre and Mahler, 2004), few studies have evaluated PAH profiles in floodplain deposits for the purposes of historical or hydrogeomorphic evaluation. The spatial variability of PAH concentrations in core profiles collected from one floodplain site was caused by the metabolization and mobilization of previously deposited PAHs and the effects of differential sedimentation and erosion rates of the floodplain (Witter et al., 2003).

6.52.3.5 Impacts of climate change on land use

Regional climates have rarely been static over centennial or millennial timescales, so evaluations of the effects of land use on fluvial systems should consider that past and future climates might be quite different from the present. Global temperatures have risen approximately 1.0 °C from pre-industrial levels and are likely to rise another 0.5 °C between 2030 and 2052 (IPCC, 2018). Since 1850, mean air temperatures over the global land surface have risen almost twice as much as the mean global air temperature (IPPC, 2019). Many studies have been conducted on potential watershed responses to the combined effects of global environmental change and climate change (Boardman et al., 1990; Walling, 1990; Pelletier et al., 2015). This section presents a few examples to illustrate the nature of some of the work that needs to be done. The topic of climate change effects on rivers is covered by other chapters in this Treatise.

Relationships between land use and climate can be considered in at least three ways:

1. effects of land use on climate,
2. effects of global regional climates on land use, and
3. how climate and climate change affect interactions between land use and erosion and sedimentation.

Land use can affect microclimates through changes such as shading at ground level by canopy, albedo, and water budgets. Shading and albedo directly govern local energy budgets, whereas moisture drives latent heat exchanges associated with water vapor flux (evapotranspiration). Advection may geographically extend these effects to neighboring areas by the transfer of energy and moisture through air temperature and humidity. Until the mid-twentieth century, however, the effects of land use on global climate were largely dismissed on the premise that the perceived impacts of human activities were restricted to local microclimates and ineffective to the global climate system (Thornthwaite, 1956). This changed quickly, however, as the importance of atmospheric chemistry to surface warming became apparent (Plass, 1956; Landsberg, 1970). An early anthropogenic climate-change hypothesis recently extended links between anthropogenic forest clearance, atmospheric CO_2 and methane, and global warming back as much as seven millennia (Ruddiman, 2003, 2007). Subsequent studies estimated that carbon emissions generated by LU/LC change were insufficient to generate such an early rise of CO_2 (Pongratz et al., 2008, 2009) because CO_2 increased at a linear rate whereas population growth was exponential (**Fig. 6**). An explanation for this apparent discrepancy was advanced based upon formerly higher per capita land clearance rates (Ruddiman and Ellis, 2009); land use was formerly less intensive but much more geographically extensive. The use of fire and shifting agriculture was widespread, so relatively small groups of hominids could clear large areas of land. Ruddiman and Ellis (2009) suggested that slash-and-burn and other forms of extensive land use release much more carbon than their modern counterparts on a per capita basis, so atmospheric carbon increased faster than population growth as early agriculture spread.

Regional climates clearly affect land use and geomorphic processes. For example, monocrop agricultural systems that work well in mid-latitude regions generally fail in the tropics, and slash-and-burn agricultural systems do not function well in perennially moist climates. Relationships between climate and geomorphic processes have long been a topic of climatic geomorphology, which is covered elsewhere in this treatise. Several nonlinearities exist in the relationships between climate, vegetation, soil erosion, and sediment yields. Some of these relationships have been explained on a conceptual basis. For example, Langbein and Schumm (1958) presented an empirical analysis of effective precipitation and sediment yield data for the USA that indicated maximum sediment yields occur in semi-arid environments (**Fig. 7**). This implies that decreasing precipitation in a subhumid forested region that causes a shift toward prairie or desert shrubland is likely to result in an increase in sediment yields. A further decrease in precipitation to full aridity may result in decreased sediment yields. Knox (1972) presented a theoretical biogeomorphic response model in which a simple sudden increase in precipitation can generate an episodic pulse of sediment that ceases once vegetation is established (**Fig. 8**). Such non-linear responses of sediment production to a simple change in precipitation depend on pre-existing climate and vegetation conditions. Furthermore, modern sediment responses to simple climatic shifts are complicated by human activities such as agricultural practices, deforestation, wetland drainage, storage in farm ponds and reservoirs, etc.

6.52.3.6 Impacts of water transfers and allocations on fluvial systems

Changes in LU/LC may have indirect effects on rivers through regional transfers of water resources. Numerous examples have been noted where water use has altered the hydrologic regimen of rivers downstream, resulting in morphologic changes. For example, streamflow in the lower Platte River decreased dramatically after the 1930s, which led to channel narrowing and floodplain

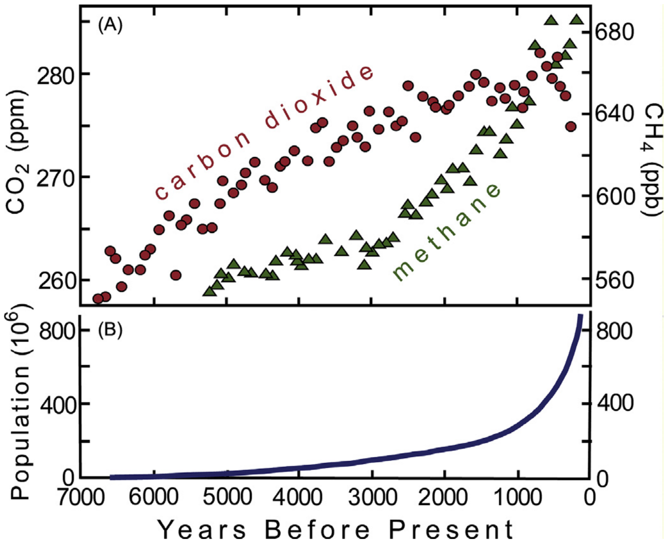

Fig. 6 Time-series plots over the past 7000 years. (a) Atmospheric CO_2 and methane increased at different rates. CO_2 increases were almost linear with time, whereas methane increased at closer to an exponential rate. (b) Population growth shows exponential growth with a strong acceleration in the past millennium. If areas of forest clearance were proportional to population, then increases in greenhouse gases were not proportional to deforestation. If, however, early hominids cleared more land per capita than later societies, then increases in greenhouse gases may have been caused by deforestation. Adapted from Ruddiman WF, Ellis EC (2009). Effect of per-capita land use changes on Holocene forest clearance and CO_2 emissions. *Quaternary Science Reviews* 28(27–28): 3011–3015.

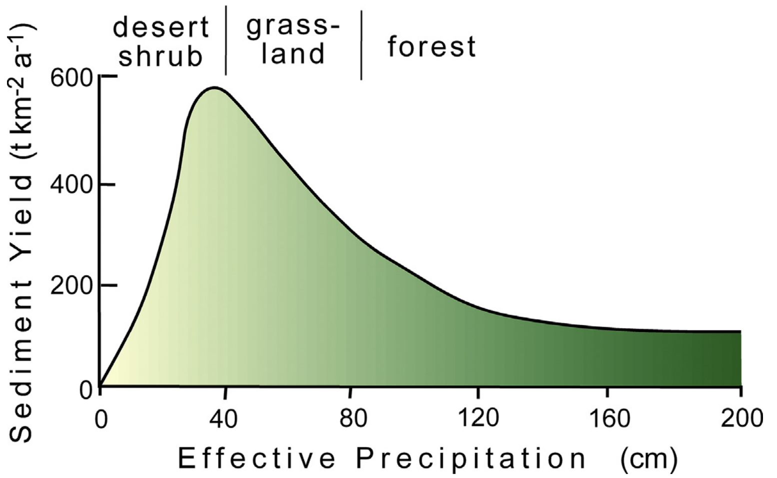

Fig. 7 Mean annual sediment yields in the USA are related to climate through interactions with land cover. Maximum yields occur in regions with small effective precipitation where vegetative cover is sparse. In more humid regions, higher precipitation is moderated by thicker vegetation protecting soils from erosion. Adapted from Langbein WB, Schumm SA (1958) Yield of sediment in relation to mean annual precipitation. *EOS, Transactions. American Geophysical Union* 39: 1076–1084, with permission from AGU.

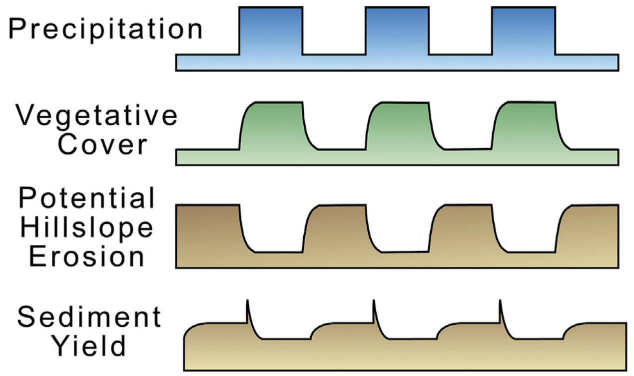

Fig. 8 The biogeomorphic response model explains how an episode of high sediment yields may be generated by a simple step-functional increase in precipitation. Time lags between moisture availability and vegetative cover and soil erodibility can be translated into an episodic increase in sediment yields with increased precipitation and a moderate increase in yields when precipitation decreases. Adapted from Knox JC (1972) Valley alluviation in Southwestern Wisconsin. *Annals of the Association of American Geographers* 62: 401–410, with permission from Association of American Geographers.

expansion (Eschner et al., 1983; Simons and Simons, 1993). By 1979, channel top widths for most of the lower river were only 8–50% of the widths recorded by surveys in the 1860s. Interbasin transfers and groundwater pumping may augment surface water supplies, enough to substantially influence low flows, mean annual flows, aquatic ecosystems, and upland LU/LC. These water transfers may support major irrigation works or municipal supply systems that result in changed land use over large areas. Large interbasin transfers in the western USA include the Los Angeles Aqueduct, the Colorado Aqueduct, the Central Valley Project of California, the California Water Project, and the Central Arizona Project (NRC, 1992). A plan to transfer 25–50 km^3 $year^{-1}$ of water south from the Ob River in Siberia to the Aral Sea was abandoned following dissolution of the former Soviet Union (Golubev and Biswas, 1984). Ironically, smaller-scale interbasin diversions via the KaraKum Canal that carried water away from the Aral Sea into the KaraKum Desert initiated regional desiccation (Micklin, 1988). Large withdrawals for irrigation between 1960 and 1990 caused the Aral Sea to drop 13 m (43 ft), lose two-thirds of its volume, and increase in salinity threefold (Clarke, 1993). The physical and environmental changes associated with desiccation of the Aral Sea and irrigation of the surrounding area are extreme (Tsytsenko, 2003). Major storage and conveyance facilities (e.g., reservoir and canal systems) may have additional impacts on fluvial systems.

Water transfers are common in small urbanizing watersheds where domestic water supplies provide water for domestic irrigation, thus increasing base flows during dry periods. Even if these changes in water supply can be critical to the management of local watersheds, water resources, and aquatic ecosystems, their direct influence on fluvial geomorphology is largely constrained to changes in vegetation that influence flood magnitudes and sediment loads. Although water resources allocations go beyond the scope of this chapter on land use, their potential should be considered for a full understanding of the extent of hydrogeomorphic influences of human activities.

6.52.4 Human impacts on fluvial systems

Fluvial systems are highly responsive to the water and sediment loadings from their watersheds. Linkages between hillslope erosion and sediment deliveries to channels can be weak or lagged in time, but high rates of upland erosion will ultimately result in downstream adjustments. The nature of these adjustments will depend, in part, on the longitudinal connectivity between uplands and fluvial systems. Given the extensive hydrologic impacts that land-use changes may have on water and sediment production, drainage systems downstream often display remarkable responses to changes in LU/LC.

6.52.4.1 Morphologic changes caused by changing flood magnitudes and sediment production

Hydrologic and sedimentologic changes induced by land use can generate substantial responses in channel morphology. These changes and their precursors have been highly variable. For example, stream restoration projects often assume that so-called 'natural' streams had meandering, single-thread channels, yet evidence indicates a great diversity of channel morphologies prior to human disturbances (Brown et al., 2018; Lewin, 2010; Notebaert et al., 2018; Walter and Merritts, 2008). Downstream morphological responses to land-use change vary greatly with landscape sensitivity, increases in runoff and sediment deliveries, and nonlinear response of sediment dynamics to anthropogenic disturbance (Notebaert and Verstraeten, 2010; Verstraeten et al., 2017). Increased storm runoff volumes and peak flows tend to enlarge channels, whereas increased sediment production tends to cause channels to aggrade. To complicate matters, both of these processes may depend on the crossing of thresholds and are time transgressive, so one part of a watershed may be responding differently from another at a given time and responses may shift downstream through time (Brown et al., 2013; Notebaert et al., 2018). Brown et al. (2013) showed that although two small catchments in the UK had a similar dramatic discontinuity in the style and rate of floodplain sedimentation that was driven by agriculture, the response was separated in time by 2300 years. These relationships are further complicated by variations in sediment texture, the partitioning of sediment between in-channel and overbank deposits, and potential armoring of the channel bed. These factors also may be time transgressive as fine-grained, in-channel deposits are rapidly delivered downstream while lag gravels and overbank deposits are slowly conveyed. For example, a major land-use change, such as deforestation or urbanization that increases water and sediment production may initially result in sediment storage and channel aggradation near the source accompanied by channel widening as coarse materials are introduced to the bed (Gomez et al., 2004). Later, headwater channels may enlarge – especially by widening if coarse materials were introduced – and downstream channels may aggrade as sediment and bed waves arrive (Gomez et al., 2003; Sims and Rutherfurd, 2017). Such a sequence may explain entrenchment of streams in small mid-Atlantic watersheds (drainage areas $< 25\ km^2$), as evidenced by greater decreases in bankfull channel recurrence frequencies than larger channels (Costa, 1975; Jacobson and Coleman, 1986; Jacobson et al., 2001). Many European studies have shown that Holocene channel pattern changes and transitions between stability and instability were related to land-use and climatic change that altered hydrological and sediment regimes (Foulds and Macklin, 2006; Lespez et al., 2008, 2015; Lewin and Macklin, 2010; Macklin et al., 2010; Morin et al., 2011; Verstraeten et al., 2017; Brown et al., 2018; Candel et al., 2018; Stutenbecker et al., 2019).

Channel enlargement can involve a combination of vertical adjustment by incision or floodplain aggradation or widening by lateral channel migration. Models of channel response to aggradation indicate that channel incision tends to evolve through a series of stages that commonly involve degradation in a narrow zone, followed by widening, then by channel-bottom aggradation to achieve a new equilibrium state (Harvey and Watson, 1986; Simon and Hupp, 1986; Simon, 1989; Lecce, 1997). Schumm (1991, 2005) presented a model of channel incision in which initial deepening is followed by widening, bank undercutting, and bank failures (**Fig. 9**). A similar model was advocated for arroyos along the Rio Puerco (Elliott et al., 1999). The remobilization of sediment by widening results in a prolonged period of high sediment flux following aggradation so that sediment waves tend to be asymmetrical with respect to time. Furthermore, channel vertical incision rates often decrease during the widening phase. Lisle and Church (2002) presented a conceptual model of decreasing sediment-transport capacity for degrading alluvial storage reservoirs consisting of two phases. The first phase is characterized by abundant sediment and a high sensitivity of local sediment storage to changes in supply. The second phase of degradation is associated with channel armoring and form roughness that impede vertical incision and lateral migration and determine spatial patterns of transport and storage.

Channel enlargement influences energy conditions that govern channel adjustment, the frequency of floodplain inundation, and the delivery of water and sediment downstream. Thus, channel enlargement in upstream areas has downstream consequences. As channels enlarge, they can contain larger flows that – if energy slopes are not decreased – may generate higher in-channel stream powers and shear stresses, leading to further incision or channel widening. These morphologic changes can decrease the downstream attenuation of flood waves, further increasing flood peaks downstream (Woltemade, 1994). Larger, more frequent overbank flows coupled with increased upland soil erosion can increase bank heights through vertical accretion on floodplain surfaces. Continued channel widening decreases flow depths for a given flow frequency, requiring larger flood magnitudes to generate a given stream power. Several studies in the Driftless Area of Wisconsin, USA, have shown that channel enlargement began in tributaries before propagating downstream (Knox, 1987; Lecce, 1997; Lecce and Pavlowsky, 2001). Channel enlargement in tributaries reduces overbank flow frequencies and floodplain sedimentation rates. This sediment is more efficiently conveyed downstream by high-energy channels that can contain larger flood flows. Thus, at the same time that tributary channels are undergoing declining sedimentation on the original floodplain surface, valleys a short distance downstream may aggrade with sediment routed through the enlarged tributary channels.

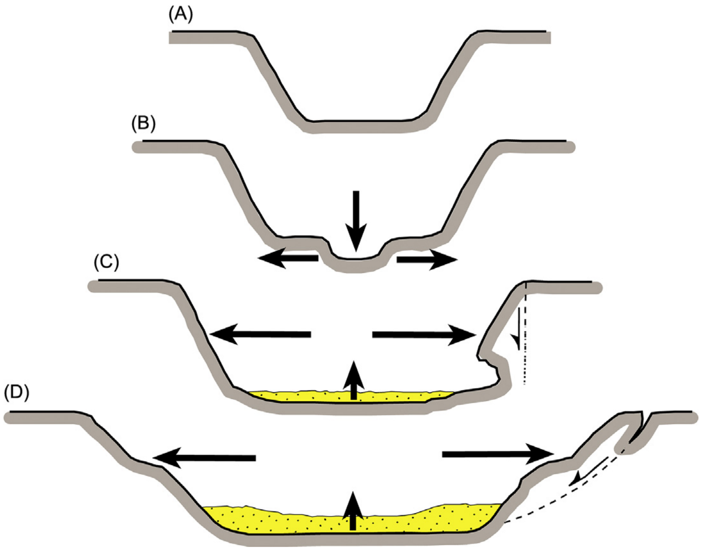

Fig. 9 Model of channel evolution in response to initial degradation (a, b), followed by bed widening, mass wasting of banks, and alluviation (c, d). Adapted with permission from Schumm SA (2005) *River Variability and Complexity*. Cambridge: Cambridge University Press, 220 pp.

Sediment deliveries are clearly related to channel bedforms, but the specific effects of the sediment must be scaled by the ability of the stream to carry its load. Linkages between sediment production and specific bedforms in mountain rivers have been identified by examining ratios of sediment transport capacity to sediment production (Montgomery and Buffington, 1997). In small mountain tributaries where transport capacities are high relative to deliveries, valley bottoms can have abundant bedrock exposures and boulder deposits and channel form is dominated by cascades (Fig. 10). Downstream within small watersheds, as sediment production increases and gradients and transport capacities decrease, bedforms transition from step pools to upper-regime plane beds. Farther downstream, where sediment supplies greatly exceed transport capacities, bedforms give way to pool and riffle sequences with dunes and ripples. This conceptualization illustrates the importance of sediment production to river form. As sediment deliveries to mountain rivers govern the nature of channel bedforms, so do changes in land use that alter these deliveries. Clearly, such a dependency is not restricted to mountain rivers. In fact, rivers may be transformed from single-thread to braided with increased sediment deliveries (Gilbert, 1917) or with an increase in the ratio of bed material to suspended material (Brice, 1982).

The lateral connectivity between channels and their floodplains can be influenced as a direct or indirect consequence of land use. Levees constructed to protect floodplains from flooding and sedimentation for urban and agricultural land uses directly reduce connectivity. Levees may cause an almost complete hydrologic isolation of the channel from its floodplain and a substantial ecologic separation (Opperman et al., 2009). Channels may also become isolated from their floodplains as an indirect consequence

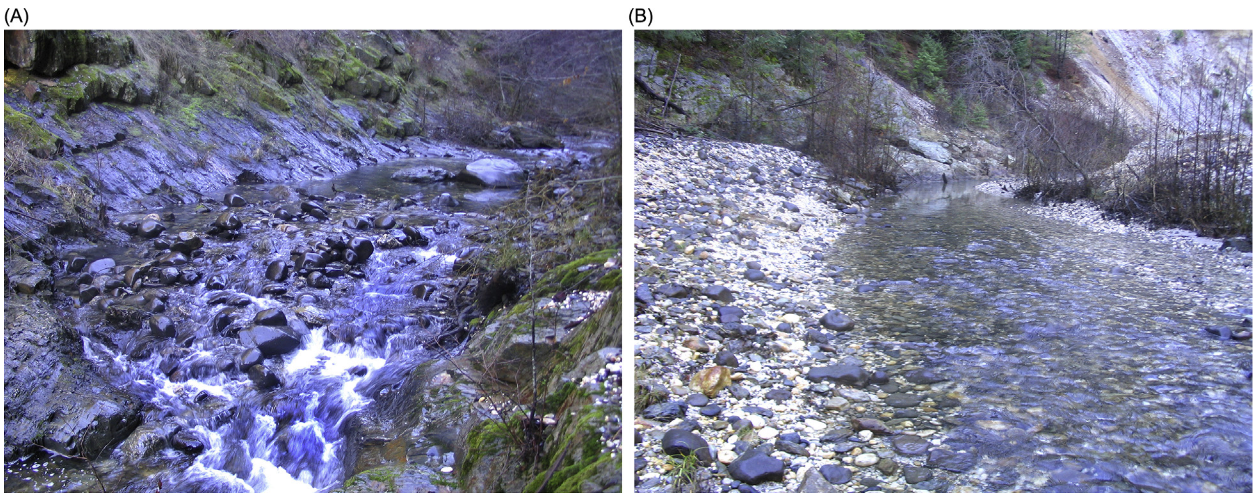

Fig. 10 Bedforms in upper Greehhorn Creek, Sierra Nevada, northern California. (A) View up cascade in bouldery bed materials. (B) View up planar gravel-bed channel B200 m downstream of first photograph below abundant supply of relatively fine gravel from hydraulic mining sediment tailings fan. Photo by LA James (December 2004).

of land use that generates an ADE. As channels aggrade, the bed rises and the frequency of floodplain inundation increases. The increased lateral connectivity may be accompanied by substantial changes to the floodplain. When aggradation ceases and degradation begins, lateral connectivity decreases, often to less than pre-aggradation conditions. The tendency for channels to vertically incise, prior to widening after an aggradation episode, was described in the previous section. The widening phase of readjustment tends to proceed more slowly (Lisle and Church, 2002), so a narrow, entrenched channel may persist for a relatively long period. In any case, the former floodplains constructed of anthropogenic sediment during the period of maximum aggradation are left as relatively high terraces (**Fig. 11**). The frequency of flooding on the high terrace surfaces and the accessibility to aquatic organisms are reduced if not eliminated.

6.52.4.2 Episodic erosion and sedimentation

Humans are environmental engineers who rearrange hydrologic and ecologic systems to garner resources and control their habitats. The resulting changes to geomorphic systems can be gradual or punctuated, and purely anthropogenic or superimposed on effects of climatic and tectonic change. Thus, the stratigraphic, sedimentologic, and geomorphic evidence of human influences can be abrupt and clear, or diffuse and obscure. In accordance with fluvial theories of thresholds and complex response (Schumm, 1977, 1979), even gradual changes may be manifested in the alluvial record as the sudden onset of a complex series of cut-and-fill features that occur in response to a disruption of system stability. Alluvial responses that appear to be gradual on contemporary timescales may also be considered episodic where observed in a longer stratigraphic record.

6.52.4.2.1 Time, episodicity, and neocatastrophism

Responses of watersheds and river systems to land-use changes can be relatively rapid when considered over long time periods, that is, decadal to millennial change (10^1–10^3 year). A considerable body of geomorphic theory concerned with rapid change can be applied to watershed responses to human activities over these timescales. Much of this theory emerged without consideration of anthropogenic processes, but can be adapted for this purpose. Testing these theories with anthropogenic applications should draw upon principles and methods of both geologic and cultural history (e.g., Brierley, 2010). Many alluvial events over the past several millennia – a period germane to discussions of intensified land use and the emergence of civilization – have been decidedly episodic. The implication of these events should be considered in light of philosophical debates over uniformitarianism versus catastrophism (Gould, 1987; Huggett, 1989, 1990). Rapid floodplain and channel changes may be characterized by an application to geomorphic systems of the concept of 'punctuated equilibria,' borrowed from evolutionary biology (Eldredge and Gould, 1972). This may seem antithetical to purely uniformitarian theories of gradualism (Playfair, 1802; Lyell, 1830; cf. Wolman and Miller, 1960; Romano, 2015). Nevertheless, it is fully compatible with modern concepts of neocatastrophism (Dury, 1980; Albritton, 1989; Huggett, 1990) that are clearly distinguished from the long-discredited religious dogma of classical catastrophism (Chorley

Fig. 11 Fluvial terraces along Greenhorn Creek, California, composed of hydraulic mining sediment. Arrow points to field assistant holding reflector. Photo by LA James (May 2007).

et al., 1964). In fact, even extreme examples of the episodicity of anthropogenically induced fluvial sedimentation pale in comparison with the bolides and Spokanian floods of neocatastrophism (Bretz, 1923, 1925; Baker, 1973). The important point is that fluvial landforms are generally formed by gradual ongoing processes, but large, rare events, such as a flood or period of severe aggradation, may also be important to morphogenesis.

To the well-established neocatastrophist principles based on processes in natural Earth systems, we must now add human agency, such as changes in land use. Humans may initiate highly effective and enduring episodes of geomorphic change and shifts in environmental and geomorphic trajectories with substantially accelerated rates of adjustment. It has been argued that uniformitarianism has severe limitations in the Anthropocene because it does not account for the important role of humans, which not only can operate at different rates but can also be entirely different than non-human systems in geologic time (Knight and Harrison, 2014). Baker (2014) disagrees with that interpretation and argues that uniformitarianism, if understood properly, remains valid and has operated across geologic Eras that were very different from one another (cf. Autin, 2016).

6.52.4.2.2 Aggradation-degradation episodes, bed waves, and sediment waves

The effects of major land-use changes on downstream river systems can occur over fairly large scales, namely, an entire drainage basin may respond over the course of decades or centuries. On these timescales, sediment produced by a large erosion event can be delivered downstream causing a period of channel aggradation, that is, a rise in the channel bed, and floodplain overbank deposition. When the high sediment production rates decrease, the bed will ultimately degrade, thus defining an *aggradation-degradation episode* (ADE) (James, 2010, 2018; James and Lecce, 2013). The well-known channel evolution model (Schumm et al., 1984; Simon and Hupp, 1986; Simon and Rinaldi, 2006; Van Dyke, 2013; Thompson et al., 2016; Phillips and Van Dyke, 2017) describes a set of possible stages that the degradation phase of an ADE may follow (James, 2018). Such episodes have been noted in many contexts including stratigraphic and historical records of events ranging from prehistoric to post-colonial periods.

The rise and fall of channel beds during an ADE has been described as a 'sediment wave,' although this term has also been used to describe a large episode of sediment flux. Some confusion surrounds the concept of sediment waves as a channel morphologic response vs. a sediment flux. Gilbert (1917), who first demonstrated sediment waves by measuring changes in channel-bed elevations through time (**Fig. 12**), described the passage of a sediment wave as analogous to the passage of a sediment hydrograph. The various uses of sediment wave terminology have been reviewed by Hoey (1992), Nicholas et al. (1995), James (2006, 2010), Lisle (2008), and Sims and Rutherfurd (2017). Related terms include bed waves, bed material waves, bedload sheets, sediment slugs, and sediment pulses (Wathen and Hoey, 1998; Lisle et al., 2001; Madej et al., 2009). Given confusion between concepts and important potential differences in timing between the bed response and the passage of sediment stored above and within a channel reach, James (2006, 2010) advocated a distinction between bed waves that are topographic changes in the bed and sediment waves that represent the passage of a mass of sediment through a reach over time (sediment flux).

Many studies have been concerned with how bed waves evolve as dispersive or translational forms. The original description by Gilbert (1917) was clearly of a series of migratory waves with down-valley translation more than 50 km. Several modern studies indicate that bed wave evolution is dominated by dispersion, although some translation also takes place (Lisle et al., 2001; Cui et al., 2003; Sklar et al., 2009). An alternative mode of response to the introduction of a large pulse of sediment is fragmentation, in which discrete bed waves translate periodically from the initial site (Gaeuman et al., 2017).

The distinction between bed waves and sediment waves allows comparisons between the timing of these responses during an ADE. Substantial changes in land use that generate massive sedimentation events may initiate an episode that is accompanied by both a bed wave and a sediment wave, although they are not necessarily synchronous. Where sediment is stored on floodplains, remobilization will likely be a slow process releasing sediment long after the channel bed has incised. In severe aggradational events generated by land-use change, deforestation, or mining, the sediment introduced to the bed may be fine grained relative to preexisting channel lag materials. In this case, as sediment deliveries decrease from their elevated rates, the channel bed will incise back to pre-aggradational levels relatively quickly (**Fig. 9**), possibly defining a relatively symmetrical bed wave. However, sediment reworking typically remains high during the prolonged period of widening, maintaining high sediment loads and producing a skewed sediment wave (**Fig. 13**). Recognizing the extended period of high sediment activity is an important dynamic to river management.

6.52.4.2.3 Legacy sediment

Alluvium deposited following human disturbances is generally referred to as 'legacy sediment' (James, 2013a). The term has been applied to postcolonial alluvium in the Americas and Oceania; however, it can also be applied to older anthropogenic alluvium as well as colluvium and human construction materials. Legacy sediment, such as post-settlement alluvium, is common on some floodplains of North America where it overlies a pre-settlement soil (Knox, 1972, 1977, 1987) (**Fig. 14**). This sediment represents sedimentation in response to episodic erosion induced by land clearance, mining, or other activities. Extensive historical alluviation occurred in the Mid-Atlantic states (Costa, 1975; Pizzuto et al., 2016), the Southeast (Happ, 1945; Trimble, 1974; Jackson et al., 2005; James, 2006; Donovan et al., 2015; Royall and Kennedy, 2016; Lyons et al., 2015; Dearman and James, 2019), the upper Midwest (Knox, 1972, 2006; Magilligan, 1985; Lecce, 1997; Faulkner, 1998; Lecce and Pavlowsky, 2001; Pavlowsky et al., 2017), and in the West (Gilbert, 1917; Singer et al., 2013; Nelson and Church, 2012). In Europe and Asia, legacy sediment was generated by multiple ADEs, resulting in a complex anthropogenic alluvial stratigraphy (Lang et al., 2003; Dotterweich, 2005; Vanwalleghem et al., 2006; Macklin and Lewin, 2008). Legacy sediment in Europe is poorly documented relative to the USA and Australia and is often obscured by reach-scale engineering projects, but may often be found behind groins and training walls (Vauclin et al., 2020). Sediment from early metal mining has been documented in the UK (Lewin et al., 1977; Macklin et al., 2006), the USA

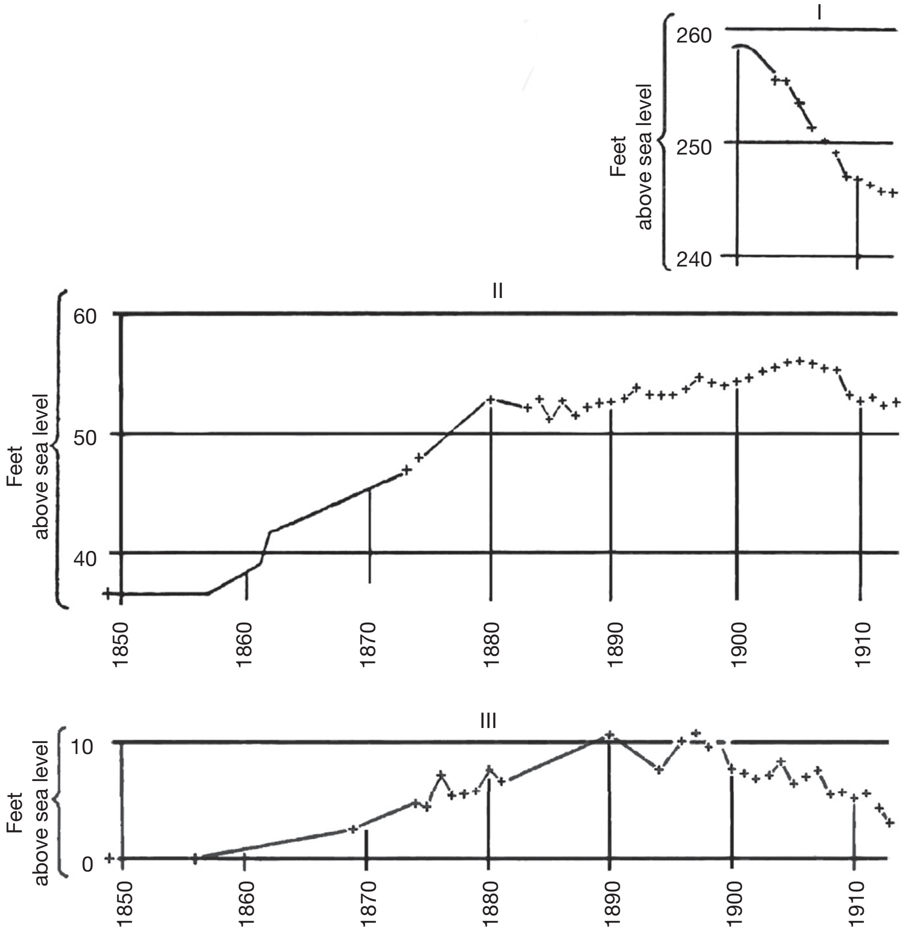

Fig. 12 Original figure by Gilbert (1917) showing bed waves during the beginning of an aggradation–degradation episode generated by hydraulic mining sediment at three locations in California. Channel beds rose 3–6 m (10–20 ft) and were degrading by 1900. (I) Lower Yuba River at the Narrows. (II) Lower Yuba River near Marysville. (III) Sacramento River at Sacramento. Reproduced from Gilbert GK (1917) *Hydraulic-mining debris in the Sierra Nevada. US Geological Survey Professional Paper 105*. Washington, DC: Government Printing Office, with permission from USGS.

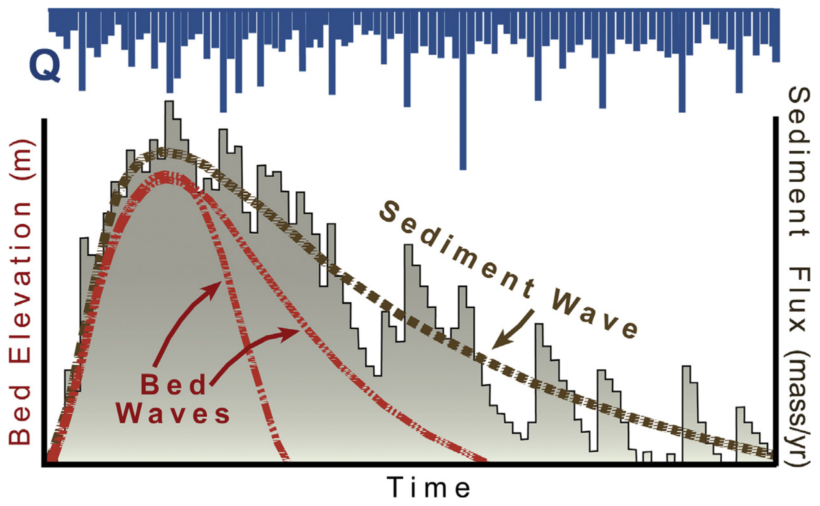

Fig. 13 Two bed waves and a skewed sediment wave. Sediment waves represent the flux of sediment and are commonly skewed with respect to time if long-term storage is involved. Bed waves represent the rise and fall of the channel bed and may be symmetrical or skewed depending on local flow hydraulics and bed sediment characteristics. Adapted from James LA (2010) Secular sediment waves, channel-bed waves, and legacy sediment. *Geography Compass*, doi: 10.1111/j.1749-8198.2010.00324.x and James LA (1999) Time and the persistence of alluvium: River engineering, fluvial geomorphology, and mining sediment in California. *Geomorphology* 31: 265–290, with permission from Wiley.

Fig. 14 Channel bank exposure of laminated historical silt and fine sand alluvium overlying a pre-settlement soil along the Blue River, Wisconsin. Photo by S Lecce.

Southeast (Leigh, 1994; Lecce et al., 2008), Midwest (Knox, 1987; Lecce and Pavlowsky, 1997), and West, including the Rocky Mountains (Hilmes and Wohl, 1995; Wohl, 2001; Dethier et al., 2018), the Sierra Nevada of California (Gilbert, 1917; James, 1989, 1991a), and Alaska (Van Haveren, 1991). Similar episodic deposition of sediment occurred in British Columbia (Nelson and Church, 2012), Australia (Brooks and Brierley, 1997) and New Zealand (Gomez et al., 2004). Wohl (2015) advocates applying the concept of legacy sediment to modern, on-going processes and expanding the definition to include not only enhanced sedimentation rates and contaminated sediment, but also to reduced sedimentation rates.

6.52.4.3 Contamination from mining and industrial pollutants

High concentrations of sediment-borne pollutants can have severe impacts on water quality and ecological diversity (MacDonald et al., 2000; Moran et al., 2017). Thus, the importance of sediment quality to river management can motivate geomorphic assessments of contaminated sediment transport and its relationship to channel form and stability (Miller, 1997; Macklin et al., 2006). Moreover, relationships between land-use patterns and fluvial landforms can be used to identify locations of contaminant inputs, storage, and remobilization in streams. Some pollutants are soluble and tend to be transported in the dissolved form, so they are only loosely linked to fluvial sediment transport and respond poorly to geomorphic controls (Salomons, 1985). However, other contaminants are typically dispersed in suspended and bed sediments that can become deposited within specific floodplain and channel features in a predictable and diagnostic manner. These pollutants may occur within the particle matrix like ore grains and coal fragments or may be strongly bound to reactive substrates on particle surfaces including clay minerals, organic matter, and iron-manganese oxides (Forstner and Wittmann, 1981; Horowitz, 1991; Kossoff et al., 2014). Sediment-associated contaminants generally include metals such as lead (Pb), zinc (Zn), copper (Cu), cadmium (Cd) and mercury (Hg), nutrients including phosphorus (P) and nitrogen (N), and hydrophobic organic compounds including some polychlorinated biphenyls (PCBs) and polycyclic aromatic hydrocarbons (PAHs).

Because of strong connections between sediments and some pollutants, contaminated sediments are often used as stratigraphic markers and transport tracers to evaluate landscape (or landform) history, transport processes, and sediment pathways in watersheds (Presley et al., 1980; Owens et al., 2001; Van Metre and Mahler, 2004). Downstream trends of metals, nutrients, and related isotopes in channel sediments and floodplain deposits are used by geomorphologists to assess the spatial and temporal impacts of land-use change on watershed hydrology, sediment sources and dispersal, and alluvial sedimentation (Miller and Orbock Miller, 2007). Conversely, interactions among geomorphic processes, sediment dynamics, and geochemical cycling can also govern flux rates and distribution of sediment-associated contaminants in aquatic environments (Martin and Meybeck, 1979; Miller and Orbock Miller, 2007; Ciszewski and Matys-Grygar, 2016). Understanding these geomorphological-geochemical interactions is important to river science and watershed management in a way that goes beyond geomorphology through identification and monitoring of pollution sources and their distribution, understanding processes of sediment and pollution transfer, and assessing environmental impacts and the ultimate fate of contaminants (Forstner, 1987; Miller, 1997; Macklin et al., 2006). Although nutrient loadings and eutrophication are important impacts of land-use change, this section focuses on fluvial metal-sediment contamination mainly from mining sources with some attention to other urban, industrial, and agricultural pollutants.

Mining-affected watersheds are attractive for geomorphic studies because: (i) mine-production records are often available to track input history, (ii) the high contrast between metal concentrations in tailings and background sediment allows the mining signal to be detected far downstream, and (iii) the particle sizes of tailings can vary significantly reflecting interactions between hydraulic processes of sedimentation of channels and floodplains (Miller, 1997; Macklin et al., 2006). Mining operations can release large amounts of tailings to river systems through uncontrolled releases, tailings dam failures, or spills (Moore and Luoma, 1990; Salomons, 1995; Gallart et al., 1999; Macklin et al., 2006; Taylor and Little, 2013; Kossoff et al., 2014; Hudson-Edwards, 2016). Fluvial processes can disperse tailings from mines several hundred kilometers downstream (Brook and Moore, 1988; Leenaers, 1989; Horowitz et al., 1990). Further, the stratigraphic signal of mining inputs typically spans 100 years or more. Indeed, some floodplain deposits contain records of mining pollution spanning 2000–4500 years BP, and therefore remain available for reworking by channel erosion or weathering to contaminate watersheds for long periods (Hudson-Edwards et al., 1999a; LeBlanc et al., 2000). For the most part, historical mining sites are available for study, being relatively numerous and distributed across a range of climate and geological regions. For example, there are 557,000 abandoned mine sites in the USA alone. Of that amount, 38,991 are on 728,434 km^2 of public lands controlled by the U.S. Forest Service with 34% having records of mineral and waste production and 3.5% involving placer mining with direct physical disturbance of stream channels and floodplains (Carr, 2005). Even Soda Butte Creek in Yellowstone National Park, Wyoming, has been contaminated by historical Pb and Cu mining (Marcus et al., 2001).

Tailings inputs can contaminate river systems in two general ways depending on the degree of geomorphic disturbance. *Passive dispersal* involves the transportation and deposition of mine wastes with little obvious geomorphic effect (Lewin and Macklin, 1987). The volume and texture of tailings inputs do not interfere substantially with the geomorphic state of the channel, but still can cause contamination. *Active transformation* involves releases of tailings to the channel in sufficient amounts to cause substantial changes in sediment composition and channel morphology (Lewin and Macklin, 1987; Moore and Luoma, 1990). Indeed, Lewin (2013) reported that historical mining disturbances, including excavations, dump piles, and tailings releases, were a causal factor in the geomorphic modification of floodplains in the UK over the past 400 years. In most situations, the texture and mineralogy of tailings differ dramatically from the natural stream load. Therefore, studies of channel systems affected by active transformation caused by extreme tailings loads or larger particle sizes can use grain counts and other bulk characteristics of the tailings materials, rather than geochemical analyses, to evaluate sediment transport, storage, and channel effects (Adams, 1944; Knighton, 1989; James, 1991b; Hudson-Edwards, 2003; Pavlowsky et al., 2017).

6.52.4.3.1 Early contributions to geochemical fluvial geomorphology

Research from different fields contributed to the development of the geomorphic-geochemical approach to river studies (Miller and Orbock Miller, 2007). Initially, geochemical studies of rivers focused on the dissolved transport phase, largely ignoring particulate or sediment-bound elements or contaminants (Forstner and Wittmann, 1981). Landscape denudation studies focused on the influences of geology and hydrology on dissolved load transport with some effort to understand the effects of land use (Meybeck, 1976; Gunn, 1982; Walling and Webb, 1983). Some of the first studies of the transport of metals and radioisotopes in association with suspended and bed sediment occurred in the early 1960s (Sayre et al., 1963; Glover, 1964; Turekian and Scott, 1967). Published studies on the elemental composition of bedrock formations and soils provided a framework to better understand the spatial variability of lithogenic contributions of metals to sediment loads at the global scale (e.g., Turekian and Wedepohl, 1961; Taylor, 1964; Shacklette and Boerngen, 1984). In the 1970s, the importance of suspended sediment and the solid phase of metal transport to geochemical balances in watersheds was recognized (Turekian and Scott, 1967; Perhac and Whelan, 1972; Gibbs, 1973, 1977; Martin and Meybeck, 1979).

Concepts from the field of exploration geochemistry greatly advanced the development of methods to separate background (undisturbed) sources from anomalous (contaminated) inputs of metals in channel sediments and to quantify the dispersal patterns of contaminants below their source (Levinson, 1974, 1980). Reach-scale variations in suspended sediment-metal transport in streams were found to be controlled by discharge, sediment supply, and pollution source characteristics (Symader and Thomas, 1978; Symader, 1980). In undisturbed small watersheds in the southwestern USA, vegetation and bedrock characteristics were strongly linked to metal concentrations and nutrients in bank deposits and to channel sediments downstream in both the sand and silt/clay fractions (Gosz et al., 1980; White and Gosz, 1983). By 1980, strong connections among land use, pollution inputs, and river sediment contamination were recognized (Forstner and Muller, 1981). Calls were made to geomorphologists to complete applied research on environmental problems concerning channel instability and sedimentation (Wolman, 1977; Graf et al., 1980). Forstner and Wittmann (1981) published the first major book covering land-use impacts on sediment-metal transport and deposition in aquatic systems.

6.52.4.3.2 Metal-sediment transport

Metal-sediment interactions tend to be relatively stable under most prevailing weathering and diagenetic conditions (Lewin et al., 1977; Graf, 1996a, b). Sediments can bind metals to much higher concentrations than in the surrounding water column and yield higher signal-noise ratios (Horowitz, 1991). For example, exploration surveys generally showed that stream-water anomalies commonly extended as much as 3 km downstream, but that sediment anomalies were detectable up to 30 km below an ore body (Levinson, 1974). Moreover, the downstream dispersal of dissolved metals was limited by physical dilution and well-buffered pH, while dispersal trends of sediment metals in the channel and floodplain extended far downstream in the Danube and Marisa drainage basins in Bulgaria (Bird et al., 2010). Natural metal enrichment of stream sediments can occur below ore bodies (Fuge et al., 1991), but mining activities can increase metal concentrations in stream sediments by up to three orders of magnitude compared to natural geochemical dispersion with mining contamination potentially extending 500 km downstream compared to only 20 km for the natural trend (Helgen and Moore, 1996). However, assumptions regarding geochemical stability in sediment studies need to be verified because metal dissolution and translocation during transport and within sediment deposits can obscure transport pathways and reduce confidence in evaluating land-use relationships (Waslenchuk, 1975; Combest, 1991). For example, age or depth relationships within floodplain deposits can be obscured by rapid sedimentation, bioturbation, changes in water-table depth, and pedogenesis (Taylor, 1996; Hudson-Edwards et al., 1998; Ciszewski et al., 2008).

Geochemical characteristics of sediments contaminated by mine tailings depend on patterns of dispersion, such as distance of transport below the mine, degree of floodplain reworking, solution effects of water chemistry (e.g., pH, dissolved oxygen), and sediment binding capacity (Lewin et al., 1977; Bradley and Cox, 1986; Macklin and Dowsett, 1989; Moore and Luoma, 1990; Horowitz, 1991). Therefore, metal concentrations are not uniformly distributed in fluvial sediments because hydraulic forces can differentially sort tailings particles and background sediment by size and density downstream, across the valley floor, and among channel units (Sear and Carver, 1996; Ladd et al., 1998; Pavlowsky et al., 2010, 2017; Szabo et al., 2020). Silts and clays tend to become stored in vertically accreting overbank deposits, sands in laterally accreting point bar deposits, and gravels in floodplain splay and channel lag deposits (Bradley and Cox, 1986; Knox, 1987; Lecce and Pavlowsky, 2001; Ciszewski and Matys-Grygar, 2016).

Grain size controls the capacity of sediments to bind, concentrate, and retain trace elements from the water column or pore water with metal concentrations typically increasing with the percentage of fine particles (e.g., < 63 μm) and organic matter in sediment (Taylor, 2007). Metal concentrations increase because of larger surface areas, cation exchange capacities, and sorption densities of fine-grained particles (Horowitz and Elrick, 1987). In general, geochemical substrates such as clay, organic matter, and iron oxides also bind metals and tend to covary in fine sediments (Combest, 1991; Miller and Orbock Miller, 2007). As weathering and diagenesis proceeds, metals may become more concentrated over time in finer sediment fractions and geochemical substrates of fluvial sediments in industrialized and urbanized rivers (Forstner and Wittmann, 1981; Combest, 1991).

In contrast to rivers where contaminants are concentrated in finer sediments, metal concentrations in mining-affected streams are often poorly correlated with grain size and organic matter because of the episodic nature of mining inputs, detrital contamination across a range of particle sizes, and a relatively short time available for geochemical dissolution and redistribution to more sorptive substrates (Bradley and Cox, 1986; Brook and Moore, 1988; Leenaers et al., 1988; Moore et al., 1989; Graf et al., 1991; Taylor, 1996; Lecce and Pavlowsky, 1997; Pavlowsky et al., 2017). The sand or fine gravel fraction may have higher contaminant concentrations than the clay fraction, because of inputs of coarse tailings (Bradley and Cox, 1986; Brook and Moore, 1988; Pavlowsky et al., 2017). Even the fine gravel fraction can be important for metal transport in mining-affected streams. For example, coarse tailings grains (2–8 mm) from historical metal sulfide mines can contain residual metal concentrations of up to 22,000 ppm Zn (Pavlowsky, 1996) and 4500 ppm Pb (Pavlowsky et al., 2017). Therefore, sediment mixing between tailings and background sediments can obscure relationships between grain size and contaminant concentrations in mining-affected streams (Marron, 1989; Moore et al., 1989).

Density sorting can also influence transport and the resulting spatial distribution of metal concentrations on a stream bed affected by mine tailing inputs. Tailings particles typically contain higher density "heavy" ore minerals including metal sulfides, oxides, and carbonates with specific gravity > 4 compared to host rock minerals with specific gravity values < 3 (Taggart, 1945; Lewin et al., 1977; Yim, 1981; Macklin and Dowsett, 1989). The *hydraulic equivalent diameter*, i.e., the grain size with the same settling velocity, can be much smaller for heavy ore-enriched particles can be than for lower-density particles (Rubey, 1933; Rittenhouse, 1943; Tourtelot, 1968). It appears that the optimum size equivalent for density sorting in fluvial sediment is in the fine to medium sand fraction (0.1–0.5 mm), and little density sorting is assumed to occur with particles smaller than coarse silt (< 0.05 mm) (Day and Fletcher, 1991). Sand-sized heavy particles can be deposited at concentrations up to eight-times background in higher energy reaches with coarser gravel beds and within areas of flow separation in the channel (Best and Brayshaw, 1985; Pavlowsky, 1996). In general, channel units that concentrate heavy particles include riffle heads, tributary junctions, shallow pools, runs, and breaks in channel slope (Yim, 1981; Fletcher et al., 1987; Day and Fletcher, 1989, 1991; Macklin and Dowsett, 1989). Longitudinal density sorting occurs when less-dense sediment tends to be preferentially transported downstream.

Where higher density particles have been introduced as tailings, the importance of the heavy mineral fraction as a source of metal pollution generally decreases downstream because of dilution and selective deposition in channel deposits (Lewin et al., 1977; Macklin and Dowsett, 1989). However, surprisingly, ore particles about 0.02 mm in diameter from tailings sources were found in reservoir deposits 500 km downstream from the mine (Horowitz et al., 1990). Dense ore grains can contain very high metal concentrations. For example, in the Upper Mississippi Valley Zinc-Lead District, Wisconsin, which mainly produced sphalerite (Zn sulfide), the average Zn concentration and heavy mineral percentage (HM) in the <2 mm fraction increased in this order: undisturbed channel sediments ($n = 31$), 161 ppm Zn and 0.6% HM; mining contaminated channel sediments ($n = 103$), 2319 ppm Zn and 1.8% HM; and dolomitic gravity mill tailings ($n = 17$), 18,357 ppm Zn and 10.7% HM (Pavlowsky, 1996). The Zn concentrations in HM concentrates averaged 13.5% for tailings ($n = 11$) and 15.5% in contaminated sediments ($n = 45$) with pure sphalerite expected to contain about 64% Zn (Pavlowsky, 1996).

6.52.4.3.3 Dilution and tributary mixing processes

The combination of chemical and hydraulic processes often produces a logarithmic decrease in metal concentrations with distance below the source caused by the dilution and mixing of contaminated sediment with increasing amounts of unpolluted sediment supplied by tributary inputs and bank erosion downstream (Lewin et al., 1977; Wolfenden and Lewin, 1978; Yim, 1981; Leenaers et al., 1988; Marcus et al., 2001; Leigh, 1997; Pavlowsky et al., 2010; Singer et al., 2013; Lecce and Pavlowsky, 2014). Area-based dilution models were first used with some success to plan sampling schemes and locate ore bodies during geochemical exploration surveys (Rose et al., 1970; Hawkes, 1976). These models describe the dilution process as a decrease in the source signal at a rate that is exponentially proportional to drainage area (Phillips, 1988). Tributary sediment yields have also been used to calculate area-weighted averages of metal concentrations from upstream and tributary inputs (Marcus, 1987, 1989).

Dilution-based models have been used to predict pre-mining dispersion trends for natural sediments and post-mining sediment metal contamination to assess environmental impacts of mining for remediation purposes (Helgen and Moore, 1996). Regression equations based on dilution modeling principles have been used to quantify the decreasing metal concentrations in stream sediments from multiple mining source points within the Galena River Watershed, Wisconsin (Pavlowsky, 1996). Regression analysis was also used to account for the effects of both watershed-scale dilution and reach-scale sedimentation on the dispersal of Hg and Cu concentrations in active bed sediments in the Little Buffalo-Dutch Buffalo Creek system below the Gold Hill mining district, North Carolina (Pavlowsky et al., 2010). Not surprisingly, dilution models have also been developed using drainage density and stream order to predict the locations and dispersion of geochemical anomalies because these network characteristics tend to scale with drainage area and hydrologic output (Carranza, 2004; Shahrestani and Mokhtari, 2017).

6.52.4.3.4 Channel sediment longitudinal dispersal

Channel sediments can be analyzed to quantify the longitudinal dispersal characteristics of mining contaminants in streams (Lewin et al., 1977). Generally, bed sediments are useful for water-quality studies and monitoring programs to identify pollution "hot-spots," establish a pollution history of an area, and provide a surrogate or alternative media for determining geochemical values for suspended sediment (Horowitz and Elrick, 2017). Bed sediments were used to assess the spatial extent of contamination from historical Pb and Ag mining within Coer d'Alene lake River-Lake system in northern Idaho (Maxfield et al., 1970; Reece et al., 1978). An early bed-sediment survey of the Buffalo River, Arkansas, found that ore metal concentrations peaked below tributaries that drain old mines (Steele and Wagner, 1975). Early research on sediment-metal concentrations, associations with mineral and geochemical substrates, and dispersal was also completed in urban and industrial watersheds (Wall et al., 1978; Dong et al., 1984).

One of the first geomorphic assessments of mining-contaminated sediment transport and deposition occurred in River Ystwyth in mid-Wales (Lewin et al., 1977). Spatial trends of Pb, Zn, Cu, and Cd were related to channel hydrology, sediment characteristics, and floodplain morphology. Lewin et al. (1977) concluded that the same mining activities in different fluvial environments were unlikely to produce similar contamination patterns and that various metals may have different patterns. For example, downstream trends over several kilometers exhibited a negative decay trend for Pb, but not for Zn because urban sources of Zn increased concentrations downstream. Attempts to use mining-related metal concentrations in bed sediments as tracers to determine sediment sources proved to be difficult because of temporal variations in sediment size, source availability, and chemical metal mobility (Lewin and Wolfenden, 1978). Wolfenden and Lewin (1978) compared downstream trends in mining-metal concentrations in stream bed sediments between Mid-Wales and the Colorado Front Range, applying a negative exponential regression curve to quantify the longitudinal dispersal pattern.

Whereas dilution processes are clearly an important explanation of downstream reductions of metal concentrations (Lewin et al., 1977; Yim, 1981), other factors can affect the spatial variability of metal concentrations in fluvial sediments. For example, departures from the expected logarithmic decline of metal concentrations in river sediments have been attributed to downstream variations of the hydraulic energy present along a stream (Graf, 1996a, b) and sediment slug-transport with little mixing by a dam break flood (Graf, 1990). Similarly, Clement et al. (2017) found no downstream trend in Pb, Cu, and As concentrations in mining sediments deposited by a catastrophic flood in the Ohinemuri River of New Zealand. They concluded from the high variance in concentrations that catchment-scale hydraulic and geomorphic factors were overridden because the mine waste was dispersed downstream by a single extreme flood, similar to that of a tailings dam failure. Locally, variations in metal concentrations tend to be related to hydrodynamic conditions and the sedimentological characteristics of the particles in transport.

A variety of fluvial processes can alter the smooth trend of decreasing sediment metal contamination downstream from sources. Deposition of plutonium-contaminated sediments in Los Alamos Canyon, New Mexico, occurred where stream power was low, hydraulic resistance was high, and geomorphic conditions provided space for storage (Graf, 1996a, b). Six fluvial attributes caused sediment transport and plutonium storage to deviate from a smooth decay trend: (i) hydraulic sorting of contaminated sediment by size; (ii) wave-like transport and uneven deposition of contaminants and background sediment in the ephemeral stream; (iii) reworking of previously contaminated deposits creating secondary sources; (iv) preferential deposition in reaches of low stream power related to local increases in valley-bottom width or bank height; (v) dilution by tributary inputs of clean sediment; and (vi) geomorphic history that differentially affected patterns of valley-floor deposition and reworking (Graf, 1996a, b; Reneau et al., 2004).

In summary, the longitudinal dispersal of sediment-borne metals from mining areas by fluvial processes generally results from interactions between three variables: rates of mine wastes discharged to the channel, the texture and mineralogy of mine wastes, and sediment transport characteristics of flows including the spatial variability of stream power and amount of floodplain reworking (Lewin et al., 1977; Bradley and Cox, 1986; Bradley, 1989; Graf, 1990; Hudson-Edwards, 2003; Bird, 2016).

6.52.4.3.5 Floodplain contaminants

Floodplains play an important role as both storage reservoirs and secondary sources of metal contaminants in mined watersheds (Bradley, 1989; Horowitz, 1991; Lecce and Pavlowsky, 1997, 2001; Coulthard and Macklin, 2003). Overbank deposits are easy to access above the waterline, less disturbed than channel-bed sediments, and temporally and spatially integrate geochemical signals from the upper watershed (Ottesen et al., 1989). Floodplain cores can yield a chemical sedimentary record of human activities with the oldest deposits or deepest strata determining background or pre-disturbance concentrations (Hindel et al., 1996; Knox, 2006). Although important for evaluating environmental history, interpretations of geochemical records in floodplain deposits need to be considered within their geomorphological context, compared with independent dating where possible, and sampled adequately to understand vertical and lateral core profile variability (Macklin et al., 1994).

Mining-metal stratigraphy and marker beds were probably first used to correlate periods of mining contamination to the age and development of floodplain features along the River Rheidol, Cardiganshire, Wales (Alloway and Davies, 1971; Davies and Lewin, 1974). In the same river, Wolfenden and Lewin (1977) contrasted the sediment texture and Pb, Zn, and Cu contamination trends between point bar and overbank deposits and described the geochemical variability they observed by the effects of episodic floodplain reworking in relation to mining outputs, sedimentary environment, and metal chemistry. Macklin (1985) related the stratigraphic "pulse" of mining-related Pb, Zn, and Cu contamination in a floodplain deposit to upstream mining history to determine sedimentation rates over a 300-year period at one site on the River Axe, Mendip, England. Knox (1987) used Zn and Pb profiles in floodplains at multiple sites to assess downstream channel response and sedimentation trends caused by agricultural land-use changes and climate-driven flood events during the past 180 years in the Galena River watershed in the Driftless Area of southwest Wisconsin. Macklin and Lewin (1989) used mining Pb and Zn metals dating and historical channel data to identify patterns of channel and floodplain adjustments in five sedimentation zones during a 115-year period for a 22 km segment of the River South Tyne, North Umbria, UK.

Lateral and vertical variations in mining-metal concentrations in alluvial deposits are generally linked to sorting by sediment size and hydro-geomorphic factors. The nature of the mining process itself can affect the pattern of downstream dispersal of tailings and metals in floodplain alluvia by determining the types of mine waste produced (Bradley and Cox, 1986) and the timing and intensity of mining-sediment production (Lewin et al., 1977; Macklin, 1985; Knox, 1987; Rang and Schouten, 1989). During floodplain formation and reworking, fluvial sorting processes tend to redistribute sediments based on grain size and channel hydraulics. In general, fine sediments accumulate through vertical overbank accretion and coarse sediments by lateral accretion as point bars or splays (Wolfenden and Lewin, 1977; Bradley and Cox, 1986; Knox, 1987; Lecce and Pavlowsky, 2001). Contamination fluctuations related to alternating high and low periods of mining activity and waste output can be preserved in point bar deposits, as well as overbank deposits (Lecce and Pavlowsky, 1997, 2001).

In addition to grain size and channel hydraulics, other geomorphic and hydrologic factors can influence the spatial patterns of floodplain contamination. Factors often cited to explain variations in floodplain contamination by mining inputs include: (i) distance of the deposit downstream from sources or away from the channel, (ii) age of the deposit, (iii) elevation of deposits relative to flood-stage frequencies during active mining, (iv) paleo-topography, and (v) rate and nature of channel activity including the date of floodplain reworking in relation to former mining (Alloway and Davies, 1971; Davies and Lewin, 1974; Leenaers and Rang, 1989; Rowan et al., 1995; Brewer and Taylor, 1997; Benito et al., 2001; Lecce and Pavlowsky, 2001; Hurkamp and Raab, 2009; Szabo et al., 2020). In some cases, metal concentrations lack obvious depositional trends because of uneven floodplain topography or tributary pulses of contaminants or clean sediment (Lewin and Wolfenden, 1978; Bradley and Cox, 1990; Lecce and Pavlowsky, 1997, 2001). In tributary channels affected by rapid overbank sedimentation or channel widening, higher bank heights can develop that decrease floodplain connectivity. The result is a "flume-like" channel that can convey flood waters and contaminated sediment downstream and progressively shift the locus of deposition to lower portions of the watershed through time (Knox, 1987; Woltemade, 1994; Lecce and Pavlowsky, 2001).

Many published studies correlate mining history to fluctuations in floodplain metal profiles to assess spatial and temporal contamination trends, date strata, and evaluate sedimentation rates. Studies from the UK include: England (Wolfenden and Lewin, 1977; Macklin, 1985; Bradley and Cox, 1986; Macklin and Lewin, 1989; Macklin et al., 1997; Hudson-Edwards et al., 1998, 1999b; Dennis et al., 2009); Wales (Davies and Lewin, 1974; Wolfenden and Lewin, 1977; Taylor, 1996; Brewer and Taylor, 1997); and

Scotland (Rowan et al., 1995). Studies from Europe include: Belgium and The Netherlands (Leenaers et al., 1988; Leenaers, 1989; Rang and Schouten, 1989; Swennen et al., 1994); Spain (Gallart et al., 1999; Benito et al., 2001; Turner et al., 2008; Adanez-Sanjuan et al., 2016); Poland (Macklin and Klimek, 1992; Ciszewski, 2003; Ciszewski and Turner, 2009; Wyzga and Ciszewski, 2010); and Slovenia (Gosar and Zibret, 2011). Studies from Oceania include: New Zealand (Clement et al., 2017), and Australia (Taylor and Little, 2013). Studies in the U.S. include: Montana (Brook and Moore, 1988; Axtmann and Luoma, 1991); South Dakota (Marron, 1989, 1992); Arizona (Graf et al., 1991); Nevada (Miller et al., 1996); Missouri (Owen et al., 2011; Pavlowsky et al., 2017); Wisconsin (Knox, 1987; Lecce and Pavlowsky, 1997, 2001); Georgia (Leigh, 1994, 1997; Wang and Leigh, 2015); North Carolina (Lecce et al., 2008; Lecce and Pavlowsky, 2014; Wang and Leigh, 2015); and Maryland (Bain and Brush, 2005).

6.52.4.3.6 Storage and remobilization of sediment-associated contaminants

Floodplains were recognized early on as an important component of the long-term dispersal process of mine tailings and metals in watersheds (Bradley, 1989). Emphasizing floodplains as a contaminant sink, Bradley and Cox (1990) developed a mining-sediment metal budget for a floodplain in a segment of the River Derwent, England, and calculated annual metal storage rates of Pb, Zn, Cu, and Cd below mining areas. Rowan et al. (1995) developed a sediment budget for the main channel of Glengonnar Water, Scotland, that emphasized floodplains as a source of Pb from historical mining and showed that channel bank erosion was the major source of metals to the stream system. Methods and assumptions for calculating sediment-metal budgets for rivers affected by mining inputs with a focus on floodplain contributions have been published (Dennis et al., 2009; Lecce and Pavlowsky, 2014; Pavlowsky et al., 2017).

The incorporation of tailings and other mining wastes within alluvial deposits effectively decreases the short-term water quality threat to downstream areas (Ongley, 1987; Macklin et al., 1997). However, the stored metals are still available for downstream transport at some later date if remobilized by physical or chemical means. Moore and Luoma (1990) envisioned a three-step process whereby the mine, mill or smelter site creates the primary sources of metal wastes including tailings, flue dust, or slag. Primary contaminants are either "stored" on site or discharged to the environment to form secondary sources, including contaminated alluvial deposits and groundwater. Tertiary sources become active when contaminated sediments are reworked and dispersed downstream or chemical remobilization releases metals by contaminated groundwater or seepage. Floodplain storage can be enhanced through increased deposition rates by breaks in slope (Graf et al., 1991 ; Bird, 2016), wider valleys (Wyzga and Ciszewski, 2010; Bird, 2016), low stream powers (Lecce and Pavlowsky, 1997), floodplain topography, such as lower banks and paleochannels (Brewer and Taylor, 1997; Lecce and Pavlowsky, 2001; Bird, 2016), and control structures (Ciszewski and Turner, 2009).

Large volumes of metalliferous mine wastes can escape on-site containment, contaminate channel and floodplain deposits for long periods, and, in some cases, cause aggradation in river valleys. Single tailings-dam failures can release millions of megagrams of mining waste and occur frequently enough to be a major concern globally (Kossoff et al., 2014). Indeed, the iron tailings spill on the Doce River in Brazil released over 35×10^6 m^3 of material and contaminated 650 km of the river with toxic metals (Hatje et al., 2017). In the North Island of New Zealand, Clement et al. (2017) estimated that $\sim 2.3 \times 10^6$ Mg of Hg, As, and cyanide contaminated mine tailings from gold mining remain stored in floodplain deposits along a 10 km segment of the Ohinemuri River. In the Black Hills, South Dakota, $\sim 42 \times 10^6$ Mg of As-rich tailings were deposited on floodplains along 40 km of Whitewood Creek below the Homestake Gold Mine (Marron, 1989, 1992). At a modern mine in Papua New Guinea, 240×10^6 Mg of tailings were deposited within 10 km of the mill outfall in the Kawerrong-Jaba River system from the active Panguna copper mine on Bougainville Island (Archer et al., 1988; Jeffery et al., 1988). In the Old Lead Belt draining the Ozark Highlands in Missouri, Pavlowsky et al. (2017) estimated that $\sim 7 \times 10^6$ Mg of dolomitic Pb sulfide tailings were released to 171 km of Big River resulting in the contamination of $\sim 144 \times 10^6$ Mg of floodplain deposits and $\sim 13 \times 10^6$ Mg in channel deposits.

Attenuation times for mining contaminant concentrations in floodplain and channel deposits to return to pre-mining levels typically range from hundreds to thousands of years (Macklin et al., 2006; Dennis et al., 2009; Bird, 2016). Floodplain storage and valley aggradation adds a long-term lag time to sediment transport in fluvial systems (Ongley, 1987; James, 1989, 1999). Natural attenuation processes need time for multiple successions of floodplain reworking, downstream dispersal of contaminated sediment, and replacement or dilution of contaminated features with cleaner sediments. Two different modeling approaches were used to evaluate the natural recovery rate of the Clark Fork River, Montana, affected by historical Cu mining for almost a century. Moore and Langner (2012) reported that it may take 200 years for sediment metal concentrations in the channel to return to near background levels. In comparison, Lauer and Parker (2008) reported that it would take >1000 years to remove all the tailings from the floodplain, much longer than one cycle of reworking of the tailings by the channel. Using a modelling technique to assess the long-term risk of contamination in the River Swale in northern England with a 200-year history of base metal mining, Coulthard and Macklin (2003) reported that $>70\%$ of the sediment contaminants will remain in the river system for more than 200 years after mine closure. Lake-core records below a mining area in Brotherswater, northwest England, indicated the minimum time required for contaminated sediment levels to fall to pre-mining levels was 54–128 years for Pb and 75–187 years for Zn (Schillereff et al., 2016).

Even after remediation, contaminated sediment will likely continue to be episodically dispersed through watersheds by floods for hundreds of years after the cessation of mining (Hudson-Edwards et al., 2003; Taylor, 2007; Bird, 2016). One consequence of the long attenuation process is that periodic reworking and weathering will progressively shift the locus of storage and new contamination problems downstream over time since metals do not decay or break down in the environment (Wolfenden and Lewin, 1977; Macklin and Lewin, 1989). Similar wave-like dispersal patterns have been observed over periods of days with radionuclide-labeled channel sediments (Glover, 1964; Sayre and Hubbell, 1965) and over centuries with large volumes of mining sediment (James, 1989; Knighton, 1989). As the attenuation process continues into the future, weathering and sorting processes may increase the proportion of bioavailable metals in redeposited sediments and thus also increase the environmental risk (Bradley and Cox, 1990; Aleksander-Kwaterczak and Helios-Rybicka, 2009; Lynch et al., 2014).

6.52.5 Historical perspective: Episodic land-use change and sediment production

The dominant causes of anthropogenic soil erosion are land clearance for farming, deforestation by grazing or silviculture, farming, and fires (Butzer, 1982; Van Andel et al., 1990; Borrelli et al., 2017). The clearance of vegetation can greatly increase soil erosion that has dramatic effects on channels. Agricultural practices can be highly erosive, have the potential to rapidly accelerate erosion and sedimentation, and can have a great influence on fluvial systems. The history of agriculture extends back more than 10 millennia (Pringle, 1998) and differs from continent to continent in timing, cultivars, technologies, and severity of erosion (Houben et al., 2009; Dotterweich, 2008; James, 2013b, 2019). The impacts of agriculture on fluvial systems, therefore, are complex and require multidisciplinary perspectives to understand fully. For example, soil erosion from the initial clearance of land during the Neolithic in Europe did not generate alluviation in major river floodplains, where sedimentation lagged behind until the technology for deep plowing arrived (Broothaerts et al., 2014; Macklin et al., 2014; Macklin and Lewin, 2018). In colluvial toe slopes and floodplains of smaller streams, however, sedimentation may have started much earlier. For example, in loess landscapes of the Wetter River in Germany, Houben et al. (2012) describe a millennial-scale delay between local colluviation caused by Neolithic agriculture (ca. 7000 cal B.P.) and the anthropogenic alluviation of floodplains (ca. 2200 B.P.).

The nature and severity of human impacts on environmental systems by indigenous cultures and colonists in the Americas recently have been debated and largely revised by geographers, anthropologists, archeologists, and paleoecologists (Butzer, 1992, 1996; Denevan, 1992; James, 2019). Assumptions that European land-use practices were introduced to pristine landscapes in the Americas and Australia, and that these practices were suddenly followed by episodic erosion, should be critically evaluated on a case-by-case basis as results can vary greatly between watersheds or regions. Some pre-European landscapes in the Americas and pre-First Fleet landscapes in Australia had strong anthropogenic imprints, and European colonization did not always generate rapid soil erosion and sedimentation responses. However, numerous examples of postcolonial ADEs have been clearly documented at many locations in the USA (James, 2019) and Australasia. Conversely, a case for substantial precolonial disturbances and relatively little disturbance associated with European colonization was advanced for New South Wales, Australia (Butzer and Helgren, 2005). On the other hand, luminescence dating of deposits has recently indicated much of the sediment considered in that study is post-settlement in age (Portenga et al., 2016). This section provides a brief introduction to the broad topics of early agricultural developments and other anthropogenic changes to the environment involving land use. It briefly describes the origins of agriculture, development of intensive Eurasian agricultural technologies, prehistoric episodes of erosion in Europe, and the relatively rapid introduction of highly erosive agricultural technologies to North America, often resulting in massive ADEs. A more detailed account is provided elsewhere in this treatise.

6.52.5.1 Early land-use change and geomorphic responses

Human impacts on Earth's surface are of such antiquity, extent, and intensity that a new geological time known as the Anthropocene Epoch has been proposed (Crutzen, 2002; Waters et al., 2016). When the Anthropocene began and whether it will be formally designated as an Epoch is under debate by the International Commission on Stratigraphy (Waters et al., 2016). Proposals have ranged from an early onset (the *PaleoAnthropocene*) with the advent of agriculture (Ruddiman, 2003; Ruddiman and Ellis, 2009; Ruddiman et al., 2015), to the post-European exchange of species, to the industrial revolution, or to increasing rates of change after World War II known as the *Great Acceleration* (Steffen et al., 2015; Zalasiewicz et al., 2015, 2019). Early land-use changes brought about by the spread of agriculture and deforestation have been a crucial component of the debate over the PaleoAnthropocene. From an ecological perspective, studies have clearly demonstrated early anthropogenic changes to the composition of plant communities through the use of fire, development of cultivars, and land clearance (Ellis et al., 2013). From a geomorphic perspective, early geomorphic changes took place primarily in the form of soil erosion and sedimentation that tended to lag behind ecological changes.

The advent and spread of agriculture comprised one of the most significant changes in human history and ultimately was associated with geomorphic change. Archeological, stratigraphic, and geochronologic studies provide evidence that substantial geomorphic responses to land-use change occurred widely in prehistory in Europe (Lang et al., 2003; Dotterweich, 2008), the UK (Macklin et al., 2010), in Mesoamerica (Beach et al., 2006, 2015), and more broadly (Macklin and Lewin, 2008). The geomorphic effectiveness of pre-European human land-use change in temperate North America is not well documented (James, 2019).

Agriculture—including the domestication of plants and animals, forest clearance, and development of related stone and bone tools—emerged in many centers at various times during the early Holocene and increased in intensity long before written documents. Knowledge of the early histories and the geomorphic impacts on fluvial systems is imperfect and must be reconstructed from stratigraphic, paleobotanical, and archeologic records (Hassan, 1979; Macphail et al., 1990; Macklin, 1995; Brown et al., 2003). Archaeological terminology for the transition to agriculture varies geographically. In Europe and Asia, agriculture was related to the Neolithic stage of cultural history, whereas in eastern North America it was related to the transition from Archaic to Woodland cultural stages. The geographic expansion of agriculture was gradual and time transgressive. For example, the appearance and expansion of agriculture occurred on most continents at different times, often as a slow transition from hunting and gathering, rather than as a sudden change (Moore, 1982; Pringle, 1998; Bollongino et al., 2013). Some genetic and isotopic studies have documented clear distinctions between the DNA and diets of Mesolithic populations with limited agriculture and Neolithic populations, even though the two populations overlapped in time and shared spaces (Skoglund et al., 2012; Guiry et al., 2016). The slow transition from hunter–gatherer to a sedentary reliance on agriculture took ~3000 years in the Near East, ~6000 years in Mexico, and 4000 years in eastern North America (Smith, 1998). The gradual transition to agriculture may have begun as early as 13,000 BP in the Near East, 10,000 BP in China, 10,000 BP in Mexico, 10,000 BP in South America, and by 7500 BP in the Rhine Basin of western Europe (Van Andel et al., 1986; Pringle, 1998; Bintliff, 2002; Lang et al., 2003). Whether this expansion occurred by means of migrations of distinct groups of farmers (the demic model) or by cultural diffusion (learning and conversion of lifestyles *in situ*) has long been debated by geoarchaeologists and anthropologists. Recent DNA studies support migration as the dominant means of early agricultural expansion in parts of Europe (Fu et al., 2012; Skoglund et al., 2012).

Early agricultural expansion in China generated substantial erosion and sedimentation because of the highly erosive loess soils and a high sensitivity to climate change. After the introduction of agriculture in the late Holocene, sediment yields to the ocean from the Huang He Basin increased by an order of magnitude (Milliman et al., 1987). Sediment yields from the Huang He Basin experienced two periods of rapid sedimentation in the last 2300 years: from 600 to 1000 BP and in the late nineteenth century (**Fig. 15**; Xu, 2003). In the early period, the combination of a drier climate and intensifying land use led to the rapid degradation of forests that protected the loess hills (Xu, 2003). When humid conditions returned after AD 1100, sedimentation rates remained high as forest regeneration was inhibited by land use driven by population growth and expanded cultivation.

In northern Europe, Neolithic forest clearance and expansion of agriculture was conducted with stone axes and fire (Darby, 1956). Pollen evidence indicates that most of northern and central Europe had been covered by thick broadleaved forest through much of Roman time. The spread of Neolithic farming was associated with local clearings around settlements where the dominant pollen shifted to weeds and grain. During the Neolithic, connections between sites of upland sediment production and channels were weak, so sediment deliveries to fluvial systems relied on large storm events to convey sediment downstream. Most agriculturally derived sediment produced in the Rhine Basin during the Neolithic remained on hillslopes until the Bronze and Early Iron Ages, when gullying was initiated and deposition began to reach lower slopes and floodplains (**Fig. 16**) (Lang et al., 2003). In some cases, small bands of Neolithic people generated severe erosion, but later during the Bronze Age, large groups were often able to maintain stable settlements for long periods without episodic erosion. Butzer (1996) noted that such settlement stability could indicate a cumulative cultural experience that selected for conservationist practices, and that rapid erosion generally followed social destabilization or out migrations.

A series of technologic developments followed the Neolithic in Afro-Eurasia including invention of the wheel, steel plow shares, and the harness, increasingly intensive use of draught animals, and ultimately, the development of heavy, wheeled plows. These technologies were introduced at various times in different regions and greatly increased the geomorphic effectiveness of agriculture. The result was the increased potential for episodic upland erosion and lowland sedimentation associated with colonization, although it generally lagged behind initial settlement. In the Near East, the agricultural technology of Mesopotamia spread to the eastern Mediterranean, and by 3000 BP agriculture in the Mediterranean region was greatly altered (Van Andel et al., 1990; Lespez, 2003). Several contemporary accounts in the region describe geomorphic processes associated with severe land use such as silty rivers and delta progradation (Chorley et al., 1964). The Romans assimilated much Mesopotamian agricultural technology from the Assyrians and Greeks and introduced these technologies across their domain, including northern and western Europe (Cunliffe, 2008).

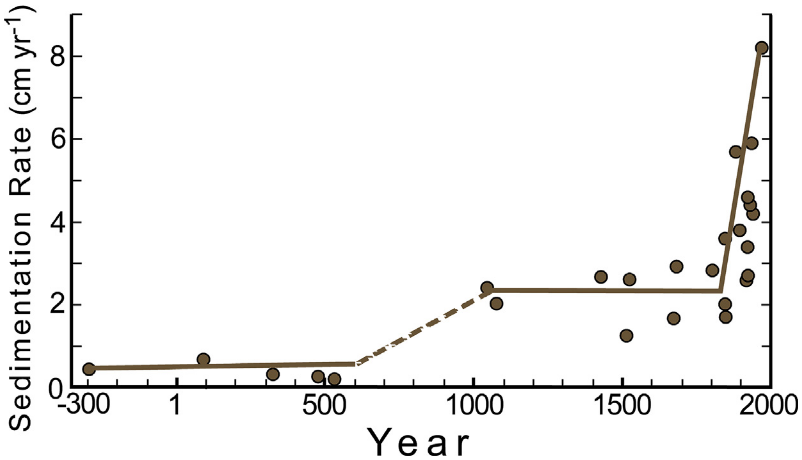

Fig. 15 Three stages of sedimentation along the Yellow River, China, showing rapid acceleration in the last 130 years. Adapted from Xu J (2003) Sedimentation rates in the lower Yellow River over the past 2300 years as influenced by human activities and climate change. *Hydrologic Processes* 17: 3359–3371, with permission from Wiley.

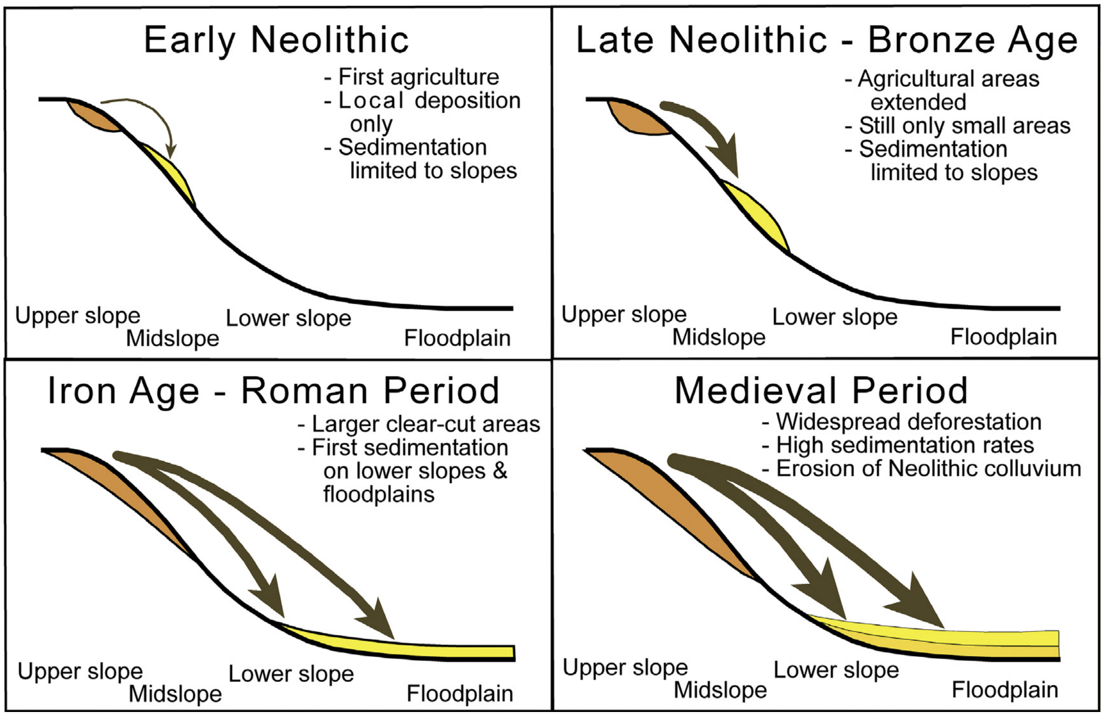

Fig. 16 Conceptual model of increasing slope–channel coupling from the early to middle agricultural period in central Europe. Adapted from Lang A, Bork H-R, Mackel R, Preston N, Wunderlich J, and Dikau R (2003) Changes in sediment flux and storage within a fluvial system: Some examples from the Rhine catchment. *Hydrologic Processes* 17: 3321–3334, with permission from Wiley.

By the Medieval period, some basins in Europe had experienced more than one episode of settlement, forest clearance, erosion, and sedimentation. For example, two periods of soil erosion and alluviation occurred in the Geul River of the Netherlands: after Roman occupation and in the Middle Ages (1000–1500 AD) (De Moor et al., 2008). Most of Germany was reforested by the mid-sixth century AD, but erosion recommenced later with renewed land clearance (Bork et al., 1998, cited in Lang et al., 2003). Rill and gully erosion became so extensive in the Medieval period (AD 1300–1700) that widespread lowland sedimentation led to the cessation of farming in some areas (Lang et al., 2003). Medieval channels and floodplains in England and Wales were transformed from highly variable, relatively complex systems to more uniform morphologies (Lewin, 2010). The introduction of heavy-wheeled plows with iron plowshares and draft animals accelerated agricultural expansion, deforestation, and deep, extensive plowing in northern and western Europe. The expansion of waterpower during the mechanical revolution led to the spread of grist and sawmills and rising exports of grain and timber. By the 1500s, forest clearance had progressed in some areas to the point that timber shortages, severe erosion, and sedimentation began to be noted (Darby, 1956). In Spain, cereal agriculture and grazing intensified during the sixteenth and nineteenth century resulting in severe erosion, although decline and land abandonment followed in the twentieth century in mountain areas (Garcia-Ruiz, 2010). By the Industrial Revolution, advanced technology–coupled with colonialism, export economies, and mercantilism–promoted the expansion of aggressive land clearance for agriculture and resource extraction overseas. As a result of these developments colonists were motivated and capable of rapid land clearance that had the potential to induce severe erosion and sedimentation.

6.52.5.2 Pre-European land use, erosion, and sedimentation in the Americas

The severity of pre- and post-European anthropogenic erosion in the Americas has been a subject of much debate over the past two decades. For most of the twentieth century, it was commonly assumed that pre-European environmental impacts in the Americas were negligible, an assumption that has been named the '*Pristine Myth*' of environmentally benign occupation (Denevan, 1992). Challenges to this assumption have documented extensive pre-European anthropogenic environmental changes in the Americas, including intensive land management and erosive land use (Butzer, 1996). Pre-European populations were much greater than early estimates but declined greatly after 1492 in response to the introduction of European diseases (Denevan, 1992). Rapid population declines after 500 BP and subsequent forest regeneration may explain a reduction in biomass burning noted in the tropical Americas that represents carbon sequestration by the biosphere of 5–10 Gt C or 2% of the global atmospheric CO_2 (Nevle and Bird, 2008). Early agriculture at many times and places in the Americas was technologically sophisticated, especially with regard to manipulating field cropping and irrigation systems, and was capable of making major environmental changes (Doolittle, 2000; Whitmore and Turner II, 2002; Denevan, 2003). For the purpose of this chapter, it is important to distinguish between ecological and geomorphic change caused by human activities (James, 2013b, 2019). Pre-European human impacts to flora and fauna in the Americas were often substantial. However, anthropogeomorphic changes in the form of accelerated erosion and sedimentation were less extensive. In some locations, however, agricultural land clearance was intense and involved terraces, irrigation canals, and mounded or ridged fields.

A 4000-year record of erosion based on sediment cores from Lake Pátzcuaro in central Mexico documents two periods of accelerated sedimentation: the Maya late Preclassic/early Classic periods (2500–1200 years BP) and the later Postclassic period (850–350 years BP) (O'Hara et al., 1993). O'Hara et al. (1993) concluded that erosion rates during the Late Preclassic/early Classic and Postclassic periods were at least as high as rates after the Spanish conquest. This indicates that erosion after the Spanish introduction of plowing was no more than that produced by traditional indigenous agricultural methods. Based on stratigraphy in dated cores and exposures, Fisher et al. (2003) argued that pre-European erosion rates in the Lake Pátzcuaro region were inversely related to population densities. Later, intense post-European erosion resulted not from the introduction of European agriculture, but from population decline and abandonment and lack of maintenance of pre-existing humanized landscapes, such as extensive terracing systems. It has been shown that neglect of agricultural terraces greatly increases the probability of their failure (Bellin et al., 2009). In the tropical lowlands of Central America, deforestation was initiated by the Maya around 4500 BP and intensified from 3500 to 3300 BP (Pohl et al., 1996; Rosenmeier et al., 2002; Beach et al., 2015). In the Maya Lowlands of Belize, buried soils from the Archaic through Preclassic Maya periods (ca. 6950 to 1750 BP) indicating geomorphic stability (Luzzadder-Beach, 2009). These paleosols are often overlain by an organic-rich clay known as the Maya clay, which contains cultivar pollen and phosphorous enrichment indicating geomorphic instability and aggradation (Beach et al., 2006; Luzzadder-Beach, 2009).

Erosional episodes have not been documented in many physiographic regions of North America. For example, evidence for high pre-European erosion rates sufficient to generate substantial aggradation-degradation episodes in temperate mid-latitude areas of most of the USA has been limited to a relatively small number of studies (Stinchcomb et al., 2011, 2013; Dotterweich, 2013; Dotterweich et al., 2014; James, 2019). Evidence of widespread pre-European agricultural practices in many parts of temperate North America is compelling. Populations were much greater than previously supposed and the use of fire was presumably effective in clearing land. Accounts of Spanish explorers include descriptions of extensive agricultural developments such as observations by Hernando de Soto's party of extensive fields and stockpiles of maize, beans, and squash in what is now the southeastern USA (Doolittle, 1992, 2000). Denevan (1992) argued that the most pristine period for the Americas was not the time of initial contact with colonists in the early sixteenth century, but two centuries later around 1750 after decimation of Indian populations. Nevertheless, stratigraphic records of extensive early anthropogenic erosion and sedimentation are rare in temperate North America. In the southeastern and midwestern USA, deep historical alluvium commonly rests abruptly over well-developed floodplain soils, indicating stable fluvial environments prior to European colonization (Happ, 1945; Knox, 1972, 1977; James, 1989; Dearman and James, 2019). Where this occurs, however, an abandoned humanized landscape hypothesis should be tested, rather than assuming no previous anthropogeomorphic change had occurred (James, 2019). Buried floodplain soils with distinct A horizons do not necessarily represent a very long period of geomorphic stability prior to European settlement. In some cases, weakly developed floodplain soils with simple A/C profiles may represent only a brief period of a few centuries of pedogenesis following land abandonment by indigenous people.

A research challenge for geomorphologists and geoarcheologists is to test the hypothesis that indigenous cultures were geomorphically effective, that is, their land clearance and land-use practices could be highly erosive on a basin-wide scale. On one hand, major urban centers such as Cahokia present evidence of high population densities and intense agricultural land use (Munoz et al., 2014). On the other hand, the geomorphic effectiveness of pre-European agriculture in the Americas was limited by the lack of animal power or the wheel for plowing or hauling freight (Goudie, 2019). By comparison, the intensive land use associated with European agriculture was potentially more geomorphically effective and had the capacity to generate rapid soil erosion and sediment redistributions from hillslopes to small watersheds.

6.52.5.3 Introduction of intensive agriculture to the colonies

Several questions have been raised about the geomorphic effectiveness of initial European colonization in the Americas. A common assumption that European colonization invariably was followed quickly by severe erosion has been referred to as the "myth of the devastated Colonial landscape" (Butzer, 1992, 1996). For example, a debate emerged in the 1990s over the degree of land degradation caused by the introduction of grazing animals to the Valle del Mezquital, Mexico, by early Spanish settlers. One pervasive theory postulated that sheep introduced by the Spaniards destabilized vegetation, resulting in a period of rapid erosion (Melville, 1994). Conversely, based on stratigraphic and sedimentologic criteria in a nearby valley, coupled with knowledge of careful Spanish livestock management practices, Butzer (1996) largely dismissed the degradational impacts of early Spanish livestock grazing. In fact, he described a sustainable, well-managed Mediterranean agricultural system in Europe that had maintained productivity for 200 generations and was not highly destructive upon its introduction in a simplified form to Mexico and Central America. Only in the eighteenth century did accelerated soil erosion and alluviation appear in those regions (Butzer, 1996). Thus, presumptions that wherever European settlement occurred in the Americas or Oceania rapid soil erosion and sedimentation necessarily followed should be critically evaluated for each watershed based on geoarcheological and stratigraphic evidence. Conversely, where agricultural practices based on prior environmental experience are introduced to a different physiographic environment, severe damage can ensue (Butzer, 1996). For instance, farming by the Incas in Peru and Ecuador utilized erosion-control practices such as terraces on steep slopes and floodplains that were maintained up to the time of Spanish conquest. Their subsequent failure, provoked by lack of maintenance and trampling by introduced grazing animals, led to severe erosion (Gade, 1992; Troeh et al., 2003). In spite of steep slopes, erosion in the former Inca region of northwestern South America was mitigated by the maintenance of indigenous land-use practices that were gradually merged with selected Spanish practices, and by the geographic isolation of the region that impeded export of lumber and other products by sea (Gade, 1992). Several studies have concluded that peak erosion rates in prehistory did not occur during times of peak population, but tended to occur after population decline and abandonment of soil conservation works such as terraces (e.g., Fisher et al., 2003; Arnaez et al., 2015).

The rapid introduction of land clearance and advanced agricultural techniques to several regions in North America and Australasia during Anglo-European colonization drastically accelerated erosion and induced unprecedented sedimentation (Happ, 1945; Knox, 1972; Brierley et al., 2005). Agricultural technology and land-use practices introduced by colonists in the USA spread rapidly westward with the frontier. In many cases, frontier settlers cleared land, worked it for a few generations until soil erosion decreased its value, and then moved farther west to clear new lands. Little attention was paid to soil conservation by frontier farmers, so erosion was often severe. By the 1930s, $\sim$1.14 million km^2 of land in the USA had been ruined or seriously damaged and another 3.14 million km^2 had experienced substantial erosion (Bennett and Lowdermilk, 1938; Bennett, 1939). The total land area of the USA at that time was 7.70 million km^2 so 15% of the total land area was seriously damaged and another 41% was substantially eroded. Erosion was especially extensive in the southern Piedmont of the USA. By 1945, Happ (1945) concluded that most Piedmont floodplains were covered with postcolonial anthropogenic sediment. All but $\sim$10% of this sediment remained in small watersheds (Trimble, 1974). Based on comparisons of suspended sediment yields with volumes of legacy sediment stored on floodplains of a small Georgia Piedmont, Jackson et al. (2005) estimated that, at present rates of sediment transport, the stored historical alluvium will not be removed for six to ten millennia. In addition, fluvial systems were substantially changed by the extirpation of beavers (Wohl, 2001).

The nineteenth-century European colonization of Australia swiftly introduced advanced agricultural technology, land clearance practices, and grazing animals that increased erosion and sediment yields. In some cases, sediment yields increased by a factor of more than 150 (Verstraeten and Prosser, 2008) and channels widened up to 340% (Brooks and Brierley, 1997). In most Australian river basins, sheet, rill, and gully erosion dominated sediment production (Olley et al., 1993; Saxton et al., 2012; Shellberg et al., 2016). Overgrazing in small watersheds often initiated gullying. Sediment production in the upper Murrumbidgee River increased by a factor of $\sim$150 compared to a twofold increase in sediment production that could have been induced by precipitation variability alone (Olley and Wasson, 2003). Widespread fluvial responses to grazing and land clearance associated with European settlement, including bank erosion and lateral migration, also extended to subtropical rivers of Queensland, Australia (Kemp et al., 2015).

Land-use intensification in many regions began late because of distance to markets or climatic or physiographic conditions that are marginal for agriculture. After World War II, major agricultural transformations occurred in response to the introduction of mechanized farm equipment, synthetic fertilizers and pesticides, availability of fossil fuels for pumping irrigation water, and development of high-yield cultivars. Agriculture spread rapidly, often into areas that had not previously been farmed extensively. The geographic spread and intensification of land clearance, cultivation, mining, and construction ultimately resulted in a tremendous increase in sediment production (Cooper et al., 2018). The question is whether soil conservation and near-site sediment retention practices will be enough to keep up with intensifying land-use developments aimed at increasing food production or urbanization. To some degree, intensive land use has shifted from land clearance for agriculture to urbanization and this poses a new set of problems and strategies for mitigation.

6.52.6 Conclusion

Anthropogenic land-use changes have had extensive effects on fluvial systems. A thorough understanding of geomorphic responses to land-use change requires both an understanding of processes and a historical perspective. The process mechanics of erosion and sedimentation are fairly well understood at the local scale, but a greater understanding is needed of how human agency alters these processes and what happens when these processes are broadened to regional and global scales. Interactions between land-use change and alterations in flood regimes, soil erosion, and sediment yields are an area of active research, especially at catchment and larger scales. Anthropogenic activities often alter sediment production, storage, and remobilization from theoretical relationships. Theoretical relationships, such as landscape sensitivity, effects of spatial scale, and forecasting climate-change impacts, need to be modified to include the effects of present and future changes to land use. One challenge is for planning, forecasting, and modeling efforts to be developed and verified on humanized landscapes. Another challenge is to upscale local studies of geomorphic change in anthropic environmental systems to global and regional scales. Yet another challenge is to integrate the results of urban studies of erosion, sedimentation, and channel morphologic change to broader applications where urban development processes occur diffusely, beyond urban centers.

Human land use clearly can generate hydrologic and geomorphic responses in the form of increased runoff, soil erosion, and sedimentation that can have substantial effects on fluvial systems. In most cases, the initial geomorphic effects of soil erosion are sedimentary deposits that tend to remain near the site. Later, that sediment may work its way downstream to larger valleys and may involve channel morphogenesis. In some cases, the geomorphic effects of agriculture may lag behind land-use alterations as long as conservation works, such as terrace systems, are maintained.

To define environmental trajectories, change must be placed in the context of antecedent conditions, such as earlier perturbations from which systems are responding. Accurate interpretations of modern geomorphic responses to land use require an understanding of past land-use changes and trends in system responses. Geomorphic impacts of land use can be traced back to the advent of agriculture, but the potential effectiveness of humans as hydrogeomorphic agents has grown greatly with technological developments and population growth. Modern changes may occur as a palimpsest on previous anthropogenic change. For example, the effectiveness of pre-European agricultural practices in the Americas has recently been revised to acknowledge potentially substantial environmental changes prior to European contact. In some regions, pre-European soil erosion was abundant but, in other regions, evidence is lacking. The hypothesis of pre-European aggradation-degradation episodes remains to be tested in many regions of North America and Oceania, based on stratigraphic evidence. Greatly accelerated rates of soil erosion and sedimentation immediately following the introduction of agriculture by European colonialists have also been questioned. Colonization did not always generate an episodic response in geomorphic systems. High spatial and temporal variability is to be expected, so the hypothesis of rapid post-European settlement alluviation should be tested in each region.

Collectively, hydrogeomorphic responses to global-scale land-use changes comprise one of the most pervasive anthropogenic changes to the environments of Earth. Many regions experienced major aggradation-degradation episodes. Recognition of past events and the legacies that they left behind, understanding the likely behavior of changed systems, and protecting against the potential for new episodes are challenges facing global-change studies. A serious commitment to land-change science is needed by geomorphologists, along with current efforts to understand the effects of climate change.

References

Adams, C., 1944. Mine wastes as a source of Galena River bed sediment. Journal of Geology 52 (4), 275–282.

Adanez-Sanjuan, P., Flem, B., Llamas, B., et al., 2016. Application of lead isotopic methods to the study of the anthropogenic lead provenance in Spanish overbank floodplain deposits. Environmental Geochemistry and Health 38, 449–468.

Alberti, M., 2005. The effects of urban patterns on ecosystem function. International Regional Science Review 28 (2), 168–192.

Albritton Jr., C.C., 1980. The Abyss of Time: Changing Conceptions of the Earth's Antiquity after the Sixteenth Century. Freeman, Cooper, San Francisco, CA, 251 pp.

Albritton Jr., C.C., 1989. Catastrophic Episodes in Earth History. Chapman and Hall, London and New York.

Aleksander-Kwaterczak, U., Helios-Rybicka, E., 2009. Contaminated sediments as a potential source of Zn, Pb, and Cd for a river system in the historical metalliferous ore mining and smelting industry area of South Poland. Journal of Soils and Sediments 9, 13–22.

Allan, J.D., 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35, 257–284.

Allison, R.J., Thomas, D.S.G., 1993. The sensitivity of landscapes. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, New York, NY, pp. 1–5.

Alloway, B.J., Davies, B.E., 1971. Trace element content of soils affected by base metal mining in Wales. Geoderma 5, 197–208.

Anderson, D.E., Goudie, A., Parker, A.G., 2013. Global Environments through the Quaternary: Exploring Environmental Change, 2nd. edn. Oxford University Press, Oxford. 424 pp.

Archer, I.M., Marshman, N.A., Salomons, W., 1988. Development of a revegetation programme for copper and sulphide-bearing mining wastes in the sub-tropics. In: Salomons, W., Forstner, U. (Eds.), Management of Solid Wastes. Springer-Verlag, Berlin, pp. 166–184.

Arnaez, J., Lana-Renault, N., Lasanta, T., Ruiz-Flano, P., Castroviejo, J., 2015. Effects of farming terraces on hydrological and geomorphological processes. A review. Catena 128, 122–134. https://doi.org/10.1016/j.catena.2015.01.021.

Arnold, C.L., Gibbons, J., 1996. Impervious surface coverage: The emergence of a key environmental indicator. Journal of the American Planning Association 62 (2), 243–258.

Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and assessment, part I: Model development. Journal of the American Water Resources Association 34 (1), 73–89.

Autin, W.J., 2016. Multiple dichotomies of the Anthropocene. The Anthropocene Review 3 (3), 218–230. https://doi.org/10.1177/2053019616646133.

Axtmann, E.V., Luoma, S.N., 1991. Large-scale distribution of metal contamination in the fine-grained sediments of the Clark Fork River, Montana, U.S.A. Applied Geochemistry 6, 75–88.

Bain, D.J., Brush, G.S., 2005. Early chromite mining and agricultural clearance: Opportunities for the investigation of agricultural sediment dynamics in the eastern Piedmont (USA). American Journal of Science 305, 957–981.

Bain, D.J., Yesilonis, I.S., Pouyat, R.V., 2012. Metal concentrations in urban riparian sediments along an urbanization gradient. Biogeochemistry 107, 67–79.

Baker, V.R., 1973. Paleohydrology and sedimentology of Lake Missoula flooding in eastern Washington. Geological Society of America Special, Paper 144, 1–79.

Baker, V.R., 2014. Uniformitarianism, earth system science, and geology. Anthropocene 5, 76–79.

Beach, T., Dunning, N., Luzzadder-Beach, S., Lohse, J., Cook, D., 2006. Ancient Maya impacts on soils and soil erosion. Catena 66 (2), 166–178.

Beach, T., Luzzadder-Beach, S., Cook, D., et al., 2015. Ancient Maya impacts on the Earth's surface: An early Anthropocene analog? Quaternary Science Reviews 124, 1–30.

Bellin, N., van Wesemael, B., Meerkerk, A., Vanacker, V., Barbera, G.G., 2009. Abandonment of soil and water conservation structures in Mediterranean ecosystems: A case study from south east Spain. Catena 76, 114–121.

Benito, G., Benito-Calvo, A., Gallart, F., et al., 2001. Hydrological and geomorphological criteria to evaluate the dispersion risk of water sludge generated by the Aznalcollar mine spill (SW Spain). Environmental Geology 40 (4–5), 417–428.

Bennett, H.H., 1939. Soil Conservation. McGraw-Hill, New York, NY, 993 pp.

Bennett, H.H., Lowdermilk, W.C., 1938. General aspects of the soil erosion problem. In: Soils and Men: Yearbook of Agriculture 1938. US Government Printing Office, Washington, DC, pp. 581–608.

Benson, M.A., 1964. Factors affecting the occurrence of floods in the Southwest. In: US Geological Survey Water Supply Paper 1580-D, 72 pp.

Best, J.L., Brayshaw, A.C., 1985. Flow separation—A physical process for the concentration of heavy minerals within alluvial channels. Journal of the Geological Society of London 142, 747–755.

Bintliff, J., 2002. Time, process and catastrophism in the study of Mediterranean alluvial history: A review. World Archaeology 33 (3), 417–435.

Bird, G., 2016. The influence of the scale of mining activity and mine site remediation on the contamination legacy of historical metal mining activity. Environmental Science and Pollution Research 23, 23456–23466.

Bird, G., Brewer, P.A., Macklin, M.G., et al., 2010. Dispersal of contaminant metals in the mining-affected Danube and Maritsa drainage basins, Bulgaria, Eastern Europe. Water, Air, and Soil Pollution. 206, 105–127.

Blauch, G.A., Jefferson, A.J., 2019. If a tree falls in an urban stream, does it stick around? Mobility, characteristics, and geomorphic influence of large wood in urban streams in northeastern Ohio, USA. Geomorphology 337, 1–14.

Bledsoe, B.P., Watson, C.C., 2001. Effects of urbanization on channel instability. Journal of the American Water Resources Association 37 (2), 255–270.

Bledsoe, B.P., Stein, E.D., Hawley, R.J., Booth, D., 2012. Framework and tool for rapid assessment of stream susceptibility to hydromodification. Journal of American Water Resources Association 48 (4), 788–808.

Boardman, J.R., 1993. The sensitivity of Downland arable land to erosion by water. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, New York, NY, pp. 211–228.

Boardman, J., Evans, R., Favis-Mortlock, D.T., Harris, T.M., 1990. Climate change and soil erosion on agricultural land in England and Wales. Land Degradation and Rehabilitation 2, 95–106.

Bollongino, R., Nehlich, O., Richards, M.P., et al., 2013. 2000 years of parallel societies in stone age central Europe. Science 342 (6157), 479–481.

Bonan, G., 1997. Effects of land use on the climate of the United States. Climatic Change 37, 449–486.

Booth, D.B., 1990. Stream-channel incision following urbanization. Journal of the American Water Resources Association 26 (3), 407–417.

Booth, D.B., Bledsoe, B.P., 2009. Streams and Urbanization-Chapter 6. In: Baker, L.A. (Ed.), The Water Environment of Cities. Springer Science and Business Media.

Booth, D.B., Fischenich, C.J., 2015. A channel evolution model to guide sustainable urban stream restoration. Area 47 (4), 408–421.

Booth, D.B., Roy, A.H., Smith, B., Capps, K.A., 2016. Global perspectives on the urban stream syndrome. Freshwater Science 35 (1), 412–420.

Bork, H.R., Bork, H., Dalchow, C., et al., 1998. Landschaftsentwicklung in Mitteleuropa. Wirkungen des Menschen auf Landschaften. Klett-Perthes, Gotha, 328 pp. (in German).

Borrelli, P., Robinson, D.A., Fleischer, L.R., et al., 2017. An assessment of the global impact of 21st century land use change on soil erosion. Nature Communications 8, 2013.

Bosch, J.M., Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. Journal of Hydrology 55 (1/4), 3–23.

Bradley, S.B., 1989. Incorporation of metalliferous sediments from historic mining into floodplains. GeoJournal 19 (1), 5–14.

Bradley, S.B., Cox, J.J., 1986. Heavy metals in the Hamps and Manifold Valleys, North Staffordshire U.K.: Distribution in floodplain soils. Science of the Total Environment 50, 103–128.

Bradley, S.B., Cox, J.J., 1990. The significance of the floodplain to the cycling of metals in the River Derwent catchment. Science of the Total Environment 97-98, 441–454.

Bradshaw, C.J.A., Sodhi, N.S., Peh, K.S.-H., Brook, B.W., 2007. Global evidence that deforestation amplifies flood and severity in the developing world. Global Change Biology 13 (11), 2379–2395.

Brander, K.E., Owen, K.E., Potter, K.W., 2004. Modeled impacts of development type on runoff volume and infiltration performance. Journal of the American Water Resources Association 40 (4), 961–969.

Brattebo, B.O., Booth, D.B., 2003. Long-term stormwater quantity and quality performance of permeable pavement systems. Water Research 37, 4369–4376.

Bretz, J.H., 1923. The channeled scabland of the Columbia Plateau. Journal of Geology 31, 617–649.

Bretz, J.H., 1925. The Spokane flood beyond the Channeled Scablands. Journal of Geology 33 (97–115), 236–259.

Brewer, P.A., Taylor, M.P., 1997. The spatial distribution of heavy metal contaminated sediment across terraced floodplains. Catena 30 (2–3), 229–249.

Brice, J.C., 1982. Stream channel stability assessment. In: Report FHWA/RD082/021. Federal Highway Administration, Washington, DC, 42 pp.

Brierley, G.J., 2010. Landscape memory: The imprint of the past on contemporary landscape forms and processes. Area 42 (1), 76–85.

Brierley, G.J., Fryirs, K., 2005. Geomorphology and River Management: Applications of the River Styles Framework. Blackwell, Oxford, 398 pp.

Brierley, G.J., Stankoviansky, M., 2002. Geomorphic responses to land use change: Lessons from differing landscape settings. Earth Surface Processes and Landforms 27, 339–341.

Brierley, G.J., Stankoviansky, M., 2003. Geomorphic responses to land use change. Catena 51, 173–179.

Brierley, G.J., Brooks, A.P., Fryirs, K.A., Taylor, M.A., 2005. Did humid-temperate rivers in the Old and New Worlds respond differently to clearance of riparian vegetation and removal of woody debris? Progress in Physical Geography 29, 27–49.

Brisset, E., Guiter, F., Miramont, C., et al., 2017. The overlooked human influence in historic and prehistoric floods in the European Alps. Geology 45, 347–350. https://doi.org/10.1130/G38498.1.

Brook, E.J., Moore, J.N., 1988. Particle size and chemical control of As, Cd, Cu, Fe, Mn, Ni, Pb, and Zn in bed sediments from the Clark Fork River, Montana (USA). Science of the Total Environment 76, 247–266.

Brookes, A., 1994. River channel change. In: Calow, P., Petts, G.E. (Eds.), The River Handbook: Hydrological and Ecological Principles. Blackwell, Oxford, pp. 55–75.

Brooks, A.P., Brierley, G.J., 1997. Geomorphic response of lower Bega River to catchment disturbance, 1851–1926. Geomorphology 18, 291–304.

Broothaerts, N., Notebaert, B., Verstraeten, G., et al., 2014. Non-uniform and diachronous Holocene floodplain evolution: A case study from the Dilje catchment, Belgium. Journal of Quaternary Science 29, 351–360.

Brown, A.G., Petit, F., James, L.A., 2003. Archaeology and human artifacts. In: Piégay, H., Kondolf, G.M. (Eds.), Tools in Fluvial Geomorphology. Oxford University Press, Oxford, pp. 59–75 ch. 3.

Brown, A.E., Zhang, L., McMahon, T.A., et al., 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. Journal of Hydrology 310 (1–4), 28–61.

Brown, A., Toms, P., Carey, C., Rhodes, E., 2013. Geomorphology of the Anthropocene: Time-transgressive discontinuities of human-induced alluviation. Anthropocene 1, 3–13. https://doi.org/10.1016/j.ancene.2013.06.002.

Brown, A.G., Lespez, L., Sear, D.A., et al., 2018. Natural vs anthropogenic streams in Europe: History, ecology and implications for restoration, river-rewilding and riverine ecosystem services. Earth-Science Reviews 180, 185–205. https://doi.org/10.1016/j.earscirev.2018.02.001.

Brunsden, D., 1990. Tablets of stone: Towards the Ten Commandments of geomorphology. Zeitschrift fu[Dot]r Geomorphologie Supplement 79, 1–37.

Brunsden, D., 1993. Barriers to geomorphological change. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, New York, NY, pp. 7–12.

Brunsden, D., 2001. A critical assessment of the sensitivity concept in geomorphology. Catena 42, 99–123.

Brunsden, D., Thornes, J.B., 1979. Landscape sensitivity and change. Transactions of the Institute of British Geographers NS 4, 463–484.

Burcher, C.L., Benfield, E.R., 2006. Physical and biological responses of stream to suburbanization of historically agricultural watersheds. Journal of North American Benthological Society 25 (2), 356–369.

Burt, T.P., Howden, N.J.K., McDonnell, J.J., et al., 2015. Seeing the climate through the trees: Observing climate and forestry impacts on streamflow using a 60-year record. Hydrological Processes 29, 473–480.

Butzer, K.W., 1982. Archaeology as Human Ecology. Cambridge University Press, Cambridge.

Butzer, K.W., 1992. The Americas before and after 1492: An introduction to current geographical research. Annals of the Association of American Geographers 82 (3), 345–368.

Butzer, K.W., 1996. Ecology in the long view: Settlement histories, agrosystemic strategies, and ecological performance. Journal of Field Archaeology 23 (2), 141–150.

Butzer, K.W., Helgren, D.M., 2005. Livestock, land cover, and environmental history: The Tablelands of New South Wales, Australia, 1820–1920. Annals of the Association of American Geographers 95 (1), 80–111.

Candel, J.H.J., Kleinhans, M.G., Makaske, B., et al., 2018. Late Holocene channel pattern change from laterally stable to meandering caused by climate and land use changes. Earth Surface Dynamics 6, 723–741. https://doi.org/10.5194/esurf-2018-31.

Carlson, T.N., Arthur, S.T., 2000. The impact of land use–land cover changes due to urbanization on surface microclimate and hydrology: A satellite perspective. Global and Planetary Change 25, 49–65.

Carr, M., 2005. A multiheaded beast: Abandoned mine lands and the challenge of water protection. Wildland Waters 4, 2–19. FS-812, Forest Service. US Department of Agriculture, Washington DC.

Carranza, E.J.M., 2004. Usefulness of stream order to detect stream sediment geochemical anomalies. Geochemistry: Exploration, Environment, Analysis 4, 341–352.

Carter, J., Owens, P.N., Walling, D.E., Leeks, G.J.L., 2003. Fingerprinting suspended sediment sources in a large urban river system. Science of the Total Environment 314-316, 513–534.
Chin, A., 2006. Urban transformation of river landscapes in a global context. Geomorphology 79, 460–487.
Chorley, R.J., Dunn, A.J., Beckinsale, R.P., 1964. The History of the Study of Landforms. Geomorphology before Davis. Methuen, London, vol. 1.
Church, M., 2017. Interpreting sediment yield scaling. Earth Surface Processes and Landforms 42, 1895–1898.
Cianfrani, C.M., Hession, W.C., Rizzo, D.M., 2006. Watershed imperviousness impacts on stream channel condition in southeastern Pennsylvania. Journal of the American Water Resources Association 42 (4), 941–956.
Ciszewski, D., 2003. Heavy metals in vertical profiles of the middle Odra River overbank sediments: Evidence for pollution changes. Water, Air, and Soil Pollution 143, 81–98.
Ciszewski, D., Matys-Grygar, T., 2016. A review of flood-related storage and remobilization of heavy metals pollutants in river systems. Water, Air, and Soil Pollution 227, 239.
Ciszewski, D., Turner, J., 2009. Storage of sediment-associated heavy metals along the channelized Odra River, Poland. Earth Surface Processes and Landforms 24, 558–572.
Ciszewski, D., Czajka, A., Blazej, S., 2008. Rapid migration of heavy metals and Cs-137 in alluvial sediments, upper Odra River valley, Poland. Environmental Geology 55, 1577–1586.
Clarke, R., 1993. Water: The International Crisis. MIT Press, Cambridge, MA, 193 pp.
Clement, A.J.H., Nováková, T., Hudson-Edwards, K.A., et al., 2017. The environmental and geomorphological impacts of historical gold mining in the Ohinemuri and Waihou river catchments, Coromandel. New Zealand. Geomorphology. https://doi.org/10.1016/j.geomorph.2017.06.011.
Combest, K.B., 1991. Trace metals in sediment: Spatial trends and sorption processes. Journal of the American Water Resources Association 27 (1), 19–28.
Cooper, A.H., Brown, T.J., Price, S.J., et al., 2018. Humans are the most significant global geomorphological driving force of the 21st century. Anthropocene Review 5 (3), 222–229.
Costa, J.E., 1975. Effects of agriculture on erosion and sedimentation in the Piedmont Province, Maryland. Geological Society of America Bulletin 86, 1281–1286.
Costa, J.E., O'Connor, J.E., American Geophysical Union, Washington, DC, 1995. Geomorphically effective floods. In: Costa, J.E., Miller, A.J., Potter, K.W., Wilcock, P.R. (Eds.), Natural and Anthropogenic Influences in Fluvial Geomorphology, AGU Geophysical Monograph Series, vol. 89, pp. 45–56.
Coulthard, T.J., Macklin, M.G., 2003. Modeling long-term contamination in river systems from historical metal mining. Geology 31 (5), 451–454.
Crutzen, P.J., 2002. Geology of mankind – the Anthropocene. Nature 415, 23.
Cui, Y., Parker, G., Lisle, T.E., Gott, J., Hansler-Ball, M.E., Pizzuto, J.E., Allmendinger, N.E., Reed, J.M., 2003. Sediment pulses in mountain rivers: 1. Experiments. Water Resource Research 39 (9), 1239. https://doi.org/10.1029/2002WR001803.
Cunliffe, B., 2008. Europe between the Oceans: Themes and Variations: 9000 BCAD 1000. Yale University Press, New Haven and London, 518 pp.
Darby, H.C., 1956. The clearing of the woodland in Europe. In: Thomas Jr., W.L., Sauer, C.O., Bates, M., Mumford, L. (Eds.), Man's Role in Changing the Face of the Earth, vol. 1. University of Chicago Press, Chicago, pp. 183–216.
Davies, B.E., Lewin, J., 1974. Chronosequences in alluvial soils with special reference to historic lead pollution in Cardiganshire, Wales. Environmental Pollution 6, 49–57.
Day, S.J., Fletcher, W.K., 1989. Effects of valley and local channel morphology on the distribution of gold in stream sediments from Harris Creek, British Columbia, Canada. Journal of Geochemical Exploration 32, 1–16.
Day, S.J., Fletcher, W.K., 1991. Concentrations of magnetite and gold at bar and reach scales in a gavel bed stream, British Columbia, Canada. Journal of Sedimentary Petrology 61 (6), 871–882.
De Moor, J.J.W., Kasse, C., van Balen, R., et al., 2008. Human and climate impact on catchment development during the Holocene: The Geul River, the Netherlands. Geomorphology 98, 316–339.
de Vente, J., Poesen, J., 2005. Predicting soil erosion and sediment yield at the basin scale: Scale issues and semi-quantitative models. Earth-Science Reviews 71, 95–125.
de Vente, J., Poesen, J., Arabkhedri, M., Verstraeten, G., 2007. The sediment delivery problem revisited. Progress in Physical Geography 31 (2), 155–178.
de Vente, J., Poesen, J., Verstraeten, G., et al., 2013. Predicting soil erosion and sediment yield at regional scales: Where do we stand? Earth-Science Reviews 127, 16–29.
Dearman, T., James, L.A., 2019. Patterns of legacy sediment deposits in a small South Carolina Piedmont catchment, USA. Geomorphology 343, 1–14.
Dedkov, A., 2004. The relationship between sediment yield and drainage basin area. In: Golosov, V., Belyaev, V., Walling, D.E. (Eds.), Sediment Transfer Through the Fluvial System, International Association of Hydrological Sciences Publication 288. IAHS Press, Wallingford, pp. 197–204.
DeFries, R., Eshleman, K.N., 2004. Land-use change and hydrologic processes: A major focus for the future. Hydrological Processes 18 (11), 2183–2186.
Dendy, F.E., Bolton, G.C., 1976. Sediment yield runoff-drainage area relationships in the United States. Journal of Soil and Water Conservation 31, 264–266.
Denevan, W.M., 1992. The pristine myth: The landscape of the Americas in 1492. Annals of the Association of American Geographers 82 (3), 369–385.
Denevan, W.M., 2003. Cultivated Landscapes of Native Amazonia and the Andes. Oxford University Press, New York, NY, 396 pp.
Dennis, I.A., Coulthard, T.J., Brewer, P., Macklin, M.G., 2009. The role of floodplains in attenuating contaminated sediment fluxes in formerly mined drainage basins. Earth Surface Processes and Landforms 34, 453–466.
Dethier, D.P., Ouimet, W.B., Murphy, S.F., et al., 2018. Anthropocene landscape change and the legacy of nineteenth- and twentieth-century mining in the Fourmile Catchment, Colorado Front Range. Annals of the American Association of Geographers 108, 917–937.
Dijkstra, J.J., Comans, R.N.J., Schokker, J., van der Meulen, M.J., 2019. The geological significance of novel anthropogenic materials: Deposits of industrial waste and by-products. Anthropocene 28, 100229.
Diodato, N., Grauso, S., 2009. An improved correlation model for sediment delivery ratio assessment. Environmental Earth Sciences 59 (1), 223–231.
Dong, A., Chesters, G., Simsiman, G.V., 1984. Metal composition of soil, sediments, and urban dust and dirt samples from the Menomonee River watershed, Wisconsin, U.S.A. Water, Air, and Soil Pollution 22, 257–275.
Donovan, M., Miller, A., Baker, M., Gellis, A., 2015. Sediment contributions from floodplains and legacy sediments to Piedmont streams of Baltimore County, Maryland. Geomorphology 235, 88–105.
Doolittle, W.E., 1992. Agriculture in North America on the eve of contact: A reassessment. Annals of the Association of American Geographers 82 (3), 386–401.
Doolittle, W.E., 2000. Cultivated Landscapes of Native North America. Oxford University Press, New York, NY.
Dotterweich, M., 2005. High resolution chronology of a 1300 year old gully system in Northern Bavaria, Germany. Modelling longterm human-induced landscape evolution. Holocene 15, 994–1005.
Dotterweich, M., 2008. The history of soil erosion and fluvial deposits in small catchments of central Europe: Deciphering the long-term interaction between humans and the environment – A review. Geomorphology 101, 192–208.
Dotterweich, M., 2013. The history of human-induced soil erosion: Geomorphic legacies, early descriptions and research, and the development of soil conservation—A global synopsis. Geomorphology 201, 1–34.
Dotterweich, M., Ivester, A.H., Hanson, P.R., et al., 2014. Natural and human-induced prehistoric and historical soil erosion and landscape development in Southwestern Tennessee, USA. Anthropocene 8, 6–24.
Douglas, I., 1967. Man, vegetation and the sediment yields of rivers. Nature 215, 925–928.
Douglas, I., 1985. Urban Sedimentology. Progress in Physical Geography 9, 255–280.
Downs, P.W., Gregory, K.J., 1993. The sensitivity of river channels in the landscape system. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, Chichester, pp. 15–30.
Downs, P.W., Piegay, H., 2019. Catchment-scale cumulative impact of human activities on river channels in the late Anthropocene: Implications, limitations, prospect. Geomorphology 338, 88–104.

Doyle, M.W., Harbor, J.M., Rich, C.F., Spacie, A., 2000. Examining the effects of urbanization on streams using indicators of geomorphic stability. Physical Geography 21 (2), 155–181.

Dury, G.H., 1980. Neocatastrophism? A further look. Progress in Physical Geography 4, 391–413.

Dusar, B., Verstraeten, G., Koen D'Haen, K., et al., 2012. Sensitivity of the Eastern Mediterranean geomorphic system towards environmental change during the Late Holocene: A chronological perspective. Journal of Quaternary Science 27 (4), 371–382.

Eisenbies, M.H., Aust, W.M., Burger, J.A., Adams, M.B., 2007. Forest operations, extreme flooding events, and considerations for hydrologic modeling in the Appalachians – a review. Forest Ecology and Management 242 (2–3), 77–98.

Eldredge, N., Gould, S.J., 1972. Speciation and punctuated equilibrium: An alternative to phyletic gradualism. In: Schopf, T.J. (Ed.), Models in Paleobiology. Freeman, Cooper, San Francisco, CA, pp. 82–115.

Elliott, J.G., Gellis, A.C., Aby, S.B., 1999. Evolution of arroyos: Incised channels of the southwestern United States. In: Darby, S.E., Simon, A. (Eds.), Incised River Channels. Wiley, New York, NY, pp. 153–185.

Ellis, E.C., Kaplan, J.O., Fuller, D.Q., Vavrus, S., et al., 2013. Used planet: A global history. Proceedings National Academy of Science 110 (20), 7976–7985. https://doi.org/10.1073/pnas.1217241110.

Endale, D.M., Fisher, D.S., Steiner, J.L., 2006. Hydrology of a zero-order Southern Piedmont watershed through 45 years of changing agricultural land use. Part 1. Monthly and seasonal rainfall–runoff relationships. Journal of Hydrology 316 (1–4), 1–12. https://doi.org/10.1016/j.jhydrol.2005.04.008.

Eschner, T.R., Hadley, R.F., Crowley, K.D., 1983. Hydrologic and morphologic changes in channels of the Platte River Basin in Colorado, Wyoming, and Nebraska: A historical perspective. In: US Geological Survey Professional Paper 1277A.

Evans, R., 1993. Sensitivity of the British landscape to erosion. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, New York, NY, pp. 189–210.

Fankhauser, R., 1999. Automatic determination of imperviousness in urban areas from digital orthophotos. Water Science and Technology 39 (9), 81–86.

Faulkner, D.J., 1998. Spatially variable historical alluviation and channel incision in west-central Wisconsin. Annals of the Association of American Geographers 88, 666–685.

Fernández-Raga, M., Palencia, C., Keesstra, S., et al., 2017. Splash erosion: A review with unanswered questions. Earth-Science Reviews 171, 463–477.

Fisher, C., Pollard, H.P., Israde, I., et al., 2003. Reexamination of human induced environmental change within the Lake Pátzcuaro Basin, Michoacán, Mexico. Proceedings of the National Academy of Sciences 100 (8), 4957–4962.

Fitzpatrick, F.A., Peppler, M.C., 2010. Relationship of urbanization to stream habitat and geomorphic characteristics in nine metropolitan areas of the United States. In: U.S. Geological Survey Scientific Investigations Report 2010–5056.

Flanagan, D.C., Frankenberger, J.R., Ascough II, J.C., 2012. WEPP: Model use, calibration, and validation. Transactions ASABE 55 (4), 1463–1477. https://doi.org/10.13031/2013.42254.

Fletcher, W.K., Dousett, P.E., Ismail, Y.B., 1987. Elimination of hydraulic effects for cassiterite in a Malaysian stream. Journal of Geochemical Exploration 28, 385–408.

Forstner, U., 1987. Sediment-associated contaminants—an overview of scientific bases for developing remedial options. Hydrobilogia 149, 221–246.

Forstner, U., Muller, G., 1981. Concentrations of heavy metals and polycyclic aromatic hydrocarbons in river sediments: Geochemical background, man's influence, and environmental impact. Geo Journal 5 (5), 417–432.

Forstner, U., Wittmann, G.T.W., 1981. Metal Pollution in the Aquatic Environment, 2nd edn. Springer-Verlag, Berlin.

Foster, G.C., Dearing, J.A., Jones, R.T., et al., 2003. Meteorological and land use controls on past and present hydro-geomorphic processes in the pre-alpine environment: An integrated lake–catchment study at the Petit Lac d'Annecy, France. Hydrological Processes 17, 3287–3305. https://doi.org/10.1002/hyp.1387.

Foulds, S.A., Macklin, M.G., 2006. Holocene land-use change and its impact on river basin dynamics in Great Britain and Ireland. Progress in Physical Geography - Earth Environment 30, 589–604. https://doi.org/10.1177/0309133306071143.

Fryirs, K., 2013. (Dis)connectivity in catchment sediment cascades: A fresh look at the sediment delivery problem. Earth Surface Processes and Landforms 38, 30–46.

Fryirs, K.A., 2017. River sensitivity: A lost foundation concept in fluvial geomorphology. Earth Surface Processes and Landforms 42 (1), 55–70.

Fryirs, K., Brierley, G.J., 1999. Slope-channel decoupling in Wolumla catchment, N.S.W., Australia: The changing nature of sediment sources following European settlement. Catena 35, 41–63.

Fryirs, K., Brierley, G.J., 2001. Variability in sediment delivery and storage along river courses in Bega catchment, NSW, Australia: Implications for geomorphic river recovery. Geomorphology 38, 237–265.

Fryirs, K.A., Brierley, G.J., 2009. Naturalness and place in river rehabilitation. Ecology and Society 14 (1), 20. Available at. http://www.ecologyandsociety.org/vol14/iss1/art20/. (Accessed October 2010).

Fryirs, K., Spink, A., Brierley, G., 2009. Post-European settlement response gradients of river sensitivity and recovery across the upper Hunter catchment, Australia. Earth Surface Processes and Landforms 34 (7), 897–918.

Fu, Q.M., Rudan, P., Paabo, S., Krause, J., 2012. Complete mitochondrial genomes reveal Neolithic expansion into Europe. Plos One 7 (3), e32473. https://doi.org/10.1371/journal.pone.0032473.

Fuge, R., Glover, S.P., Pearce, N.J.G., Perkins, W.T., 1991. Some observations on heavy metal concentrations in soils of the Mendip region of north Somerset. Environmental Geochemistry and Health 13 (4), 193–196.

Gade, D.W., 1992. Landscape, system, and identity in the post-conquest Andes. Annals of the Association of American Geographers 82 (3), 460–477.

Gaeuman, D., Stewart, R., Schmandt, B., Pryor, C., 2017. Geomorphic response to gravel augmentation and high-flow dam release in the Trinity River, California. Earth Surface Processes and Landforms 42, 2523–2540.

Gallart, F., Benito, G., Martin-Vide, J.P., et al., 1999. Fluvial geomorphology and hydrology in the dispersal and fate of pyrite mud particles released by the Aznalcollar mine tailings spill. The Science of the Total Environment 242, 13–26.

Galster, J.C., Pazzaglia, F.J., Germanoski, D., 2008. Measuring the impact of urbanization on channel widths using historic aerial photographs and modern surveys. Journal of the American Water Resources Association 44 (4), 948–960.

Gao, B., Walter, M.T., Steenhuis, T.S., et al., 2003. Investigating ponding depth and soil detachability for a mechanistic erosion model using a simple experiment. Journal of Hydrology 277 (1–2), 116–124.

Garcia-Ruiz, J.M., 2010. The effects of land uses on soil erosion in Spain: A review. Catena 81 (1), 1–11. https://doi.org/10.1016/j.catena.2010.01.001.

Gellis, A.C., Myers, M.K., Noe, G.B., et al., 2017. Storms, channel changes, and a sediment budget for an urban-suburban stream, Difficult Run, Virginia, USA. Geomorphology 278, 128–148.

Giacomoni, M.H., Gomez, R., Berglund, E.Z., 2014. Hydrologic impact assessment of land cover change and stormwater management using the hydrologic footprint residence. Journal of the American Water Resources Association 50 (5), 1242–1256.

Gibbs, R.J., 1973. Mechanisms of trace metal transport in rivers. Science 180, 71–73.

Gibbs, R.J., 1977. Transport phases of transition metals in the Amazon and Yukon Rivers. Geological Society of America Bulletin 88, 829–843.

Gilbert, G.K., 1877. Report on the geology of the Henry Mountains. US Geographical and Geological Survey of the Rocky Mountain Region. US Government Printing Office, Washington, DC.

Gilbert, G.K., 1917. Hydraulic-mining debris in the Sierra Nevada. In: US Geological Survey Professional Paper 105. Government Printing Office, Washington, DC.

Gillespie, J.L., Noe, G.B., Hupp, et al., 2018. Floodplain trapping and cycling compared to the streambank erosion of sediment and nutrients in an agricultural watershed. Journal of the American Water Resources Association 54 (2), 565–582.

Glacken, C.J., 1967. Traces on the Rhodian Shore: Nature and Culture in Western Thought from Ancient Times to the End of the Eighteenth Century. University California Press, London, 713 pp.

Glover, R.E., 1964. Dispersion of dissolved or suspended materials in flowing streams. In: US Geological Survey Professional Paper 433-B. Government Printing Office, Washington, D.C.
Goetz, S.J., Wright, R.K., Smith, A.J., et al., 2003. IKONOS imagery for resource management: Tree cover, impervious surfaces, and riparian buffer analyses in the mid-Atlantic region. Remote Sensing of Environment 88, 195–208.
Golubev, G.N., Biswas, A.K., 1984. Large-scale water transfer: Emerging environmental and social issues. International Journal of Water Resources Development 2 (2–3), 1–5.
Gomez, B., Banbury, K., Marden, M., et al., 2003. Gully erosion and sediment production in a low-order basin: Te Weraroa Stream, North Island, New Zealand. Water Resources Research 39 (7), 1187. https://doi.org/10.1029/2002WR001342.
Gomez, B., Brackley, H.L., Hicks, D.M., et al., 2004. Organic carbon in floodplain alluvium: Signature of historic variations in erosion processes associated with deforestation, Waipaoa River basin, New Zealand. Journal of Geophysical Research 109, F04011. https://doi.org/10.1029/2004JF000154.
Goodchild, M.F., Quattrochi, D.A., 1997. Scale, Multiscaling, Remote Sensing and GIS. Scale in Remote Sensing and GIS. CRC Press, Boca Raton, FL.
Gosar, M., Zibret, G., 2011. Mercury contents n the vertical profiles through alluvial sediment as a reflection of mining in the Idrija (Slovenia). Jornal of Geochemical Exploration 110 (2), 81–91.
Gosz, R.R., White, C.S., Folliott, P.F., 1980. Nutrient and heavy metal transport capabilities of sediment in the southwestern United States. Journal of the American Water Resources Association 16 (5), 927–933.
Goudie, A.S., 2019. The Human Impact on the Natural Environment: Past, Present, Future, 8th edn. J. Wiley & Sons, Hoboken, NJ, USA.
Goudie, A.S., 2020. The Human Impact in geomorphology – 50 years of change. Geomorphology 366, 106601.
Gould, S.J., 1987. Time's Arrow, Time's Cycle: Myth and Metaphor in the Discovery of Geological Time. Harvard University Press, Cambridge, MA.
Graf, W.L., 1975. The impact of suburbanization on fluvial geomorphology. Water Resources Research 11 (5), 690–692.
Graf, W.L., 1990. Fluvial dynamics of thorium-230 in the Church Rock Event, Puerco River, New Mexico. Annals of the Association of American Geographers 80 (3), 327–342.
Graf, W.L., 1996a. Fluvial geomorphic analysis of plutonium-contaminated sediment transport and deposition in Los Alamos Canyon, New Mexico. Geological Society of America Bulletin 108, 1342–1355.
Graf, W.L., 1996b. Geomorphology and policy for restoration of impounded American rivers: What is "natural?". In: Rhoads, B.L., Thorn, C.E. (Eds.)The Scientific Nature of Geomorphology. Proceedings of the 27th Binghamton Symposium. 27–29 September. Wiley, New York, NY, pp. 443–473.
Graf, W.L., 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. Water Resources Research 35, 1305–1311.
Graf, W.L., Trimble, S.W., Toy, T.J., Costa, J.E., 1980. Geographic geomorphology in the eighties. The Professional Geographer 32 (3), 279–284.
Graf, W.L., Clark, S.L., Kammerer, M.T., et al., 1991. Geomorphology of heavy metals in the sediments of Queen Creek, Arizona, U.S.A. Catena 18, 567–582.
Gregory, K.J., Chin, A., 2002. Urban stream channel hazards. Area 34 (3), 312–321.
Gregory, K.J., Davis, R.J., Downs, P.W., 1992. Identification of river channel change due to urbanization. Applied Geography 12 (4), 299–318.
Guiry, E.J., Hillier, M., Boaventura, R., et al., 2016. The transition to agriculture in south-western Europe: New isotopic insights from Portugal's Atlantic coast. Antiquity 90 (351), 604–619. https://doi.org/10.15184/aqy.2016.34.
Gunn, J., 1982. Magnitude and frequency properties of dissolved solids transport. Zeitschrift fur Geomorphologie 26, 505–511.
Guy, H.P., 1970. Sediment problems in urban areas. In: Water in the Urban Environment, U.S. Geological Survey circular 601-E.
Hall, J., Arheimer, B., Borga, M., et al., 2014. Understanding flood regime changes in Europe: A state-of-the-art assessment. Hydrology and Earth System Sciences 18, 2735–2772.
Hammer, T.R., 1972. Stream channel enlargement due to urbanization. Water Resources Research 8 (6), 1530–1540.
Happ, S., 1945. Sedimentation in South Carolina Piedmont valleys. American Journal of Science 243, 113–126.
Harvey, M.D., Watson, C.C., 1986. Fluvial processes and morphological thresholds in incised channel restoration. Journal of the American Water Resources Association 22 (3), 359–368.
Hassan, F.A., 1979. Geoarchaeology: The geologist and archaeology. American Antiquity 44, 267–270.
Hatje, V., Pedreira, R.M.A., de Rezende, C.E., et al., 2017. The environmnetnal impacts of one of the largest tailing dam failures worldwide. Scientific Reports 7, 10706. https://doi.org/10.1038/s41598-017-11143-x.
Hawkes, H.E., 1976. The downstream dilution of stream sediment anomalies. Journal of Geochemical Exploration 6, 345–358.
Hawley, R.J., Bledsoe, B.P., 2013. Channel enlargement in semiarid suburbanizing watersheds: A southern California case study. Journal of Hydrology 496, 17–30.
Hawley, R.J., MacMannis, K.R., Wooten, M.S., 2013. Bed coarsening, riffle shortening, and channel enlargement in urbanizing watersheds, northern Kentucky, USA. Geomorphology 201, 111–126.
He, Q., Walling, D.E., 2003. Testing distributed soil erosion and sediment delivery models using Cs-137 measurements. Hydrological Processes 17, 901–916.
Heathman, G.C., Larose, M., Ascough II, J.C., 2009. Soil and Water Assessment Tool evaluation of soil and land use geographic information system data sets on simulated stream flow. Journal of Soil and Water Conservation 64 (1), 17–32. https://doi.org/10.2489/jswc.64.1.17.
Heimsath, A.M., Dietrich, W.E., Nishiizumi, K., Finkel, R.C., 1999. Cosmogenic nuclides, topography, and the spatial variation of soil depth. Geomorphology 27, 151–172.
Heimsath, A.M., Furbish, D.J., Dietrich, W.E., 2005. The illusion of diffusion: Field evidence for depth-dependent sediment transport. Geology 33 (12), 949–952.
Helgen, S.O., Moore, J.N., 1996. Natural background determination and impact quantification in trace metal-contaminated river sediments. Environmental Science and Technology 30 (1), 129–135.
Henshaw, P.C., Booth, D.B., 2000. Natural restabilization of stream channels in urban watersheds. Journal of the American Water Resources Association 36 (6), 1219–1236.
Hession, E.C., Pizzuto, J.E., Johnson, T.E., Horwitz, R.J., 2003. Influence of bank vegetation on channel morphology in rural and urban watersheds. Geology 31 (2), 147–150.
Hilmes, M.M., Wohl, E.E., 1995. Changes in channel morphology associated with placer mining. Physical Geography 16, 223–242.
Hindel, R., Schalich, J., De Vos, W., et al., 1996. Vertical distribution of elements in overbank sediment profiles from Belgium, Germany, and The Netherlands. Journal of Geochemical Exploration 56 (2), 105–122.
Hinderer, M., 2012. From gullies to mountain belts: A review of sediment budgets at various scales. Sedimentary Geology 280, 21–59.
Hodgson, M.E., Jensen, J.R., Tullis, J.A., et al., 2003. Synergistic use of Lidar and color aerial photography for mapping urban parcel imperviousness. Photogrammetric Engineering and Remote Sensing 69 (9), 973–980.
Hoey, T., 1992. Temporal variations in bedload transport rates and sediment storage in gravel-bed rivers. Progress in Physical Geography 16 (3), 319–338.
Hoffmann, T., 2015. Sediment residence time and connectivity in non-equilibrium and transient geomorphic systems. Earth-Science Reviews 105, 609–627.
Hoffmann, T., Erkens, G., Gerlach, R., et al., 2009. Trends and controls of Holocene floodplain sedimentation in the Rhine catchment. Catena 77, 96–106. https://doi.org/10.1016/j.catena.2008.09.002.
Hogan, D.M., Jarnagin, S.T., Loperfido, J.V., Van Ness, K., 2014. Mitigating the effects of landscape development on streams in urbanizing watersheds. Journal of the American Water Resources Association 50 (1), 163–178.
Holman-Dodds, J.K., Bradley, A.A., Potter, K.W., 2003. Evaluation of hydrologic benefits of infiltration based urban storm water management. Journal of the American Water Resources Association 39 (1), 205–215.
Horowitz, A.J., 1991. Sediment-Trace Element Chemistry. Lewis Publishers, Chelsea, MI.
Horowitz, A.J., Elrick, K.A., 1987. The relation of stream sediment surface area, grain size and composition to trace element chemistry. Applied Geochemistry 2, 437–451.
Horowitz, A.J., Elrick, K.A., 2017. The use of bed sediments in water quality studies and monitoring programs. Proceedings of the International Association of Hydrological Sciences 375, 11–17.

Horowitz, A.J., Stephens, V.C., 2008. The effects of land use on fluvial sediment chemistry for the conterminous U.S.—results from the first cycle of the NAWQA program: Trace and major elements, phosphorous, carbon, and sulfur. Science of the Total Environment 400 (1–3), 290–314.
Horowitz, A.J., Elrick, K.A., Cook, R.B., 1990. Arsenopyrite in the bank deposits of the Whitewood Creek–Belle Fourche–Cheyenne River–Lake Oahe system, South Dakota, U.S.A. Science of the Total Environment 97/98, 219–233.
Horowitz, A.J., Meybeck, M., Idlafkih, Z., Bigger, E., 1999. Variations in trace element geochemistry in the Seine River basin based on floodplain deposits and bed sediments. Hydrological Processes 13, 1329–1340.
Horton, R.E., 1933. The role of infiltration in the hydrological cycle. EOS, Transactions American Geophysical Union 14, 446–460.
Houben, P., Wunderlich, J., Schrott, L., 2009. Climate and long-term human impact on sediment fluxes in watershed systems. Geomorphology 108, 1–7.
Houben, P., Schmidt, M., Mauz, B., et al., 2012. Asynchronous Holocene colluvial and alluvial aggradation: A matter of hydrosedimentary connectivity. The Holocene 23 (4), 544–555. https://doi.org/10.1177/0959683612463105.
Houghton, R.A., 1995. Land-use change and the carbon cycle. Global Change Biology 1, 275–287.
Hudson-Edwards, K., 2003. Sources, mineralogy, chemistry, and fate of heavy metal-bearing particles in mining-affected river systems. Mineralogical Magazine 67 (2), 2050217.
Hudson-Edwards, K., 2016. Tackling mine wastes. Science Magazine 352 (6283), 288–290.
Hudson-Edwards, K.A., Macklin, M.G., Curtis, C.D., Vaughan, D.J., 1998. Chemical remobilization of contaminant metals within floodplain sediment in an incising river system: Implications for dating and chemostratigraphy. Earth Surface Processes and Landforms 23, 671–684.
Hudson-Edwards, K.A., Macklin, M.G., Taylor, M.P., 1999a. 2000 years of sediment-borne heavy metal storage in the Yorkshire Ouse basin, NE England, UK. Hydrological Processes 13, 1087–1102.
Hudson-Edwards, K.A., Schell, C., Macklin, M.G., 1999b. Mineralogy and geochemistry of alluvium contaminated by metal mining in the Rio Tinto area, southwest Spain. Applied Geochemistry 14, 1015–1030.
Hudson-Edwards, K.A., Macklin, M.G., Jamieson, H.E., et al., 2003. The impacts of tailings dam spills and clean-up operations on sediment and water quality in river systems: The Rios Agrio-Guadiamar, Aznalcolar, Spain. Applied Geochemistry 18, 221–239.
Huggett, R.J., 1988. Dissipative systems: Implications for geomorphology. Earth Surface Processes and Landforms 13, 45–49.
Huggett, R.J., 1989. Cataclysms and Earth History: The Development of Diluvialism. Oxford University Press, Oxford, 232 pp.
Huggett, R.J., 1990. Catastrophism: Systems of Earth History. Edward Arnold, London.
Hung, C.-L.J., James, L.A., Carbone, G., 2018a. Impacts of urbanization on stormflow magnitudes in small catchments in South Carolina, USA. Anthropocene 23, 17–28.
Hung, C.-L.J., James, L.A., Hodgson, M., 2018b. An automated algorithm for mapping building impervious areas from airborne LiDAR point-cloud data for flood hydrology. GIScience & Remote Sensing. https://doi.org/10.1080/15481603.2018.1452588.
Hurkamp, K., Raab, T., 2009. Two and three-dimensional quantification of lead contamination in alluvial soils of a historic mining area using field portable X-ray fluorescence (FPXRF) analysis. Geomorphology 110 (1–2), 28–36.
IPCC (Intergovernmental Panel on Climate Change), 2018. Global warming of 1.5°C. Masson-Delmotte V. In: Zhai, P., Pörtner, H.-O., et al. (Eds.), World Meteorological Organization. Geneva, Switzerland.
IPPC, 2019. Summary for Policymakers. In: Shukla, PR, Skea, J, Calvo Buendia, E, et al. (Eds.), Climate Change and Land. World Meteorological Organization, Geneva, Switzerland.
Jackson, C.R., Martin, J.K., Leigh, D.S., West, L.T., 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation 60 (6), 298–310.
Jacobson, R.B., Coleman, D.J., 1986. Stratigraphy and recent evolution of Maryland piedmont flood plains. American Journal of Science 286, 617–637.
Jacobson, R.B., Femmer, S.R., McKenney, R.A., 2001. Land-use changes and the physical habitat of streams: A review with emphasis on studies within the US Geological Survey Federal-State Cooperative Program/USGS Circular No. 1175.
James, L.A., 1989. Sustained storage and transport of hydraulic gold mining sediment in the Bear River, California. Annals of the Association of American Geographers 79, 570–592.
James, L.A., 1991a. Incision and morphological evolution of a channel recovering from hydraulic mining sedimentation. Geological Society of America Bulletin 103, 723–726.
James, L.A., 1991b. Quartz concentrations as an index of sediment mixing: Hydraulic mine tailings in the Sierra Nevada, California. Geomorphology 4, 125–144.
James, L.A., 1999. Time and the persistence of alluvium: River engineering, fluvial geomorphology, and mining sediment in California. Geomorphology 31, 265–290.
James, L.A., 2006. Bed waves at the basin scale: Implications for river management and restoration. Earth Surface Processes and Landforms 31, 1692–1706.
James, L.A., 2010. Secular sediment waves, channel-bed waves, and legacy sediment. Geography Compass. https://doi.org/10.1111/j.1749-8198.2010.00324.x.
James, L.A., 2013a. Legacy sediment: Definitions and processes of episodically produced anthropogenic sediment. Anthropocene 2, 16–26.
James, L.A., 2013b. Impacts of early agriculture and deforestation on geomorphic systems. Shroder, J. (Ed. in chief). In: James, L.A., Harden, C.P., Clague, J.J. (Eds.), Treatise on Geomorphology, Geomorphology of Human Disturbances, Climate Change, and Natural Hazards, vol. 13. Academic Press, San Diego, pp. 48–67.
James, L.A., 2018. Ten conceptual models of large-scale legacy sedimentation–a review. Geomorphology. 331, 59–77.
James, L.A., 2019. Impacts of pre- vs. postcolonial land use on floodplain sedimentation in temperate North America. Geomorphology 331, 59–77.
James, L.A., Lecce, S.A., 2013. Impacts of land-use and land-cover change on river systems. Shroder, John F. (Ed.in chief). In: Wohl, E. (Ed.), Treatise on Geomorphology, Fluvial Geomorphology, vol. 9. Academic Press, San Diego, CA, pp. 768–793.
James, L.A., Marcus, W.A., 2006. The human role in changing fluvial systems: Retrospect, inventory and prospect. Geomorphology 79, 152–171.
James, L.A., Watson, D.G., Hansen, W.F., 2007. Using Lidar to map gullies and headwater streams under forest canopy: South Carolina, USA. Catena 71, 132–144.
James, L.A., Monohan, C., Ertis, B., 2019. Long-term hydraulic mining sediment budgets: Connectivity as a management tool. Science of the Total Environment 651, 2024–2035.
Jeffery, J., Marshman, N., Salomons, W., 1988. Behavior of trace metals in a tropical river system affected by mining. In: Salomons, W., Forstner, U. (Eds.), Chemistry and Biology of Solid Waste: Dredged Material and Mine Tailings. Springer-Verlag, Berlin, pp. 259–274.
Jetten, V., de Roo, A., Favis-Mortlock, D., 1999. Evaluation of field-scale and catchment-scale soil erosion models. Catena 37, 521–541.
Jones, J.A., Grant, G.E., 1996. Peak flow responses to clearcutting and roads in small and large basins, Cascades, Oregon. Water Resources Research 32 (4), 959–974.
Judson, S., 1968. Erosion of the land, or what's happening to our continents? American Scientist 56, 356–374.
Kang, R.S., Marston, R.A., 2006. Geomorphic effects of rural-to-urban land use conversion on three streams in the Central Redbed Plains of Oklahoma. Geomorphology 79, 488–506.
Katz, R.W., Brown, B.G., 1992. Extreme events in a changing climate: Variability is more important than averages. Climatic Change 21, 289–302. https://doi.org/10.1007/BF00139728.
Kaushal, S.S., Belt, K.T., 2012. The urban watershed continuum: Evolving spatial and temporal dimensions. Urban Ecosystems 15, 409–435.
Kemp, J., Olley, J.M., Ellison, T., McMahon, J., 2015. River response to European settlement in the subtropical Brisbane River, Australia. Anthropocene 11, 48–60.
Klein, R.D., 1979. Urbanization and stream quality impairment. Water Resources Bulletin 15 (4), 948–963.
Knight, J., Harrison, S., 2014. Limitations of uniformitarianism in the Anthropocene. Anthropocene 5, 71–75.
Knighton, A.D., 1989. River adjustments to changes in sediment load: The effects of tin mining on the Ringarooma River, Tasmania, 1875-1984. Earth Surface Processes and Landforms 14, 333–359.
Knox, J.C., 1972. Valley alluviation in Southwestern Wisconsin. Annals of the Association of American Geographers 62, 401–410.
Knox, J.C., 1977. Human impacts on Wisconsin stream channels. Annals of the Association of American Geographers 77, 323–342.
Knox, J.C., 1987. Historical valley floor sedimentation in the Upper Mississippi Valley. Annals of the Association of American Geographers 77, 224–244.
Knox, J.C., 2001. Agricultural influence on landscape sensitivity in the Upper Mississippi River Valley. Catena 42, 193–224.

Knox, J.C., 2006. Floodplain sedimentation in the upper Mississippi Valley: Natural versus human accelerated. Geomorphology 79, 286–310.

Kossoff, D., Dubbin, W.E., Alfredsson, M., et al., 2014. Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. Applied Geochemistry 51, 229–245.

Ladd, S.C., Marcus, W.A., Cherry, S., 1998. Differences in trace metal concentrations among fluvial morphological units and implications for sampling. Environmental Geology 36 (3–4), 259–270.

Lambin, E.F., Baulies, X., Bockstael, N.E., et al., 2002. Land-use and land-cover change implementation strategy. In: IGBP Report No. 48 and IHDP Report No. 10, Louvainla-Neuve, Belgium.

Landsberg, H.E., 1970. Man-made climatic changes. Science 170, 1265–1268.

Lang, A., Bork, H.-R., Mäckel, R., Preston, N., Wunderlich, J., Dikau, R., 2003. Changes in sediment flux and storage within a fluvial system: Some examples from the Rhine catchment. Hydrologic Processes 17, 3321–3334.

Langbein, W.B., Schumm, S.A., 1958. Yield of sediment in relation to mean annual precipitation. EOS, Transactions American Geophysical Union 39, 1076–1084.

Larson, M.G., Booth, D.B., Morley, S.A., 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. Ecological Engineering 18 (2), 211–226.

Lassettre, N.S., Kondolf, G.M., 2012. Large woody debris in urban stream channels: Redefining the problem. River Research Applications 28, 1377–1487.

Laub, B.G., Baker, D.B., Bedsoe, B.P., Palmer, M.A., 2012. Range of variability of channel complexity in urban, restored and forested reference streams. Freshwater Biology 57 (5), 1076–1095.

Lauer, J.W., Parker, G., 2008. Modeling framework for sediment deposition, storage, and evacuation in the floodplain of a meandering river: 2. Application to the Clark Fork River, Montana. Water Resources Research 44 (4). https://doi.org/10.1029/2006WR005529.

Leavesley, G.H., Lichty, R.W., Troutman, B.M., Sindon, L.G., 1983. Precipitation runoff modeling system: User's manual. In: US Geological Survey Water Resource Investment Report 83–4238.

LeBlanc, M., Morales, J.A., Borrego, J., Elbaz-Poulichet, F., 2000. 4,500-yar-old mining pollution in southwestern Spain: Long-term implications for modern mining pollution. Economic Geology 95, 655–662.

Lecce, S.A., 1997. Spatial patterns of historical overbank sedimentation and floodplain evolution, Blue River, Wisconsin. Geomorphology 18, 265–277.

Lecce, S.A., Kotecki, E.S., 2008. The 1999 flood of the century in eastern North Carolina: Extraordinary hydro-meteorological event or human-induced catastrophe? Physical Geography 29, 101–120.

Lecce, S.A., Pavlowsky, R.T., 1997. Storage of mining-related zinc in floodplain sediments, Blue River, Wisconsin. Physical Geography 18, 424–439.

Lecce, S.A., Pavlowsky, R.T., 2001. Use of mining-contaminated sediment tracers to investigate the timing and rates of historical floodplain sedimentation. Geomorphology 38, 85–108.

Lecce, S.A., Pavlowsky, R.T., 2014. Floodplain storage of sediment contaminated by mercury and copper from historic gold mining at Gold Hill, North Carolina, USA. Geomorphology 206, 122–132.

Lecce, S.A., Pavlowsky, R.T., Schlomer, G.S., 2008. Mercury contamination of active channel sediment and floodplain deposits from historic gold mining at Gold Hill, North Carolina, USA. Environmental Geology 55, 113–121.

Leenaers, H., 1989. Downstream changes of total and partitioned concentration in the flood deposits of the River Geul (the Netherlands). GeoJournal 19 (1), 37–43.

Leenaers, H., Rang, M.C., 1989. Metal dispersal in the fluvial system of the River Geul: The role of discharge, distance to source, and floodplain geometry. In: International Association of Hydrological Sciences publication number 184, Sediment and the Environment, Proceedings of the Baltimore Symposium.

Leenaers, H., Schouten, C.J., Rang, M.C., 1988. Variability of the metal content of flood deposits. Environmental Geology and Water Science 11 (1), 95–106.

Leigh, D.S., 1994. Mercury contamination and floodplain sedimentation from former gold mines in north Georgia. Water Resources Bulletin 30 (4), 739–748.

Leigh, D.S., 1997. Mercury-tainted overbank sediment from past gold mining in north Georgia, U.S.A. Environmental Geology 30, 244–251.

Leopold, L.B., 1968. Hydrology for urban land planning – A guidebook of the hydrologic effects of urban land use. In: U.S. Geological Survey Circular 554, 18 p.

Leopold, L.B., 1973. River channel change with time: An example: Address as retiring president of the Geological Society of America, Minnesota, November 1972. Geological Society of America Bulletin 84, 1845–1860.

Lespez, L., 2003. Geomorphic responses to long-term land use changes in Eastern Macedonia (Greece). Catena 51, 181–208.

Lespez, L., Clet-Pellerin, M., Limondin-Lozouet, N., et al., 2008. Fluvial system evolution and environmental changes during the Holocene in the Mue valley (Western France). Geomorphology 98, 55–70. https://doi.org/10.1016/j.geomorph.2007.02.029.

Lespez, L., Viel, B., Rollet, A.J., Dleahaye, D., 2015. The anthropogenic nature of present-day low energy rivers in western France and implications for current restoration projects. Geomorphology 251, 64–76.

Levinson, A.A., 1974. Introduction to Exploration Geochemistry. Applied Publishing Ltd., Calgary Alberta.

Levinson, A.A., 1980. Introduction to Exploration Geochemistry 2nd Ed.-The 1980 Supplement. Applied Publishing Ltd., Calgary, Alberta.

Lewin, J., 2010. Medieval environmental impacts and feedbacks: The lowland floodplains of England and Wales. Geoarchaeology 25, 267–311.

Lewin, J., 2013. Enlightenment and the GM floodplain. Earth Surface Processes and Landforms 38, 17–28.

Lewin, J., Macklin, M.G., 1987. Metal mining and floodplain sedimentation. In: Gardiner, V. (Ed.), International Geomorphology 1986 Part 1. Wiley, Chichester, pp. 1009–1027.

Lewin, J., Macklin, M.G., 2010. Floodplain catastrophes in the UK Holocene: Messages for managing climate change. Hydrological Processes 24, 2900–2911. https://doi.org/10.1002/hyp.7704.

Lewin, J., Wolfenden, P.J., 1978. The assessment of sediment sources: A field experiment. Earth Surface Processes and Landforms 3 (2), 171–178.

Lewin, J., Davies, B.E., Wolfenden, P.J., 1977. Interactions between channel change and historic mining sediments. In: Gregory, K.J. (Ed.), River Channel Changes. Wiley, Chichester, pp. 353–367.

Li, Y., Wang, C., 2009. Impacts of urbanization on surface runoff of the Dardenne Creek watershed, St. Charles County, Missouri. Physical Geography 30 (6), 556–573.

Lisle, T.E., 2008. The evolution of sediment waves influenced by varying transport capacity in heterogeneous rivers. In: Habersack, H., Piégay, H., Rinaldi, M. (Eds.), Gravel-Bed Rivers VI: From Process Understanding to River Restoration. Elsevier, Oxford.

Lisle, T., Church, M., 2002. Sediment transport–storage relations for degrading, gravel bed channels. Water Resources Research 38 (11), 1219. https://doi.org/10.1029/2001WR001086.

Lisle, T.E., Cui, Y., Parker, G., et al., 2001. The dominance of dispersion in the evolution of bed material waves in gravel-bed rivers. Earth Surface Processes and Landforms 26, 1409–1420.

Lu, H., Moran, C.J., Sivipalan, M., 2005. A theoretical exploration of catchment-scale sediment delivery. Water Resources Research 41. https://doi.org/10.1029/2005WR4018. W09415.

Luzzadder-Beach, S.T.B., 2009. Arising from the wetlands: Mechanisms and chronology of landscape aggradation in the Northern Coastal Plain of Belize. Annals Association of American Geographers 99 (1), 1–26. https://doi.org/10.1080/00045600802458830.

Lyell, C., 1830. Principles of Geology: Being an Attempt to Explain the Former Changes of the Earth's Surface, by Reference to Causes Now in Operation, vol. 1. John Murray, London.

Lynch, S.F.L., Batty, L.C., Byrne, P., 2014. Environmental risk of metal mining contaminated river bank sediment at redox-transitional zones. Minerals 4, 52–73.

Lyons, N.J., Starek, M., Wegmann, K.W., Mitasova, H., 2015. Bank erosion of legacy sediment at the transition from vertical to lateral stream incision. Earth Surface Processes and Landforms 40 (13). https://doi.org/10.1002/esp.3753.

Ma, B., Liu, Y., Liu, X., Ma, F., Wu, F., Li, Z., 2015. Soil splash detachment and its spatial distribution under corn and soybean cover. Catena 127, 142–151.

MacDonald, D.D., Ingersoll, C.G., Berger, T.A., 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Archives of Environmental Contamination and Toxicology 39, 20–31.

Macklin, M.G., 1985. Flood-plain sedimentation in the Upper Axe Valley, Mendip, England. Transactions of British Geographers, N.S. 10, 235–244.

Macklin, M.G., 1995. Archaeology and the river environment in Britain: A prospective review. In: Barham, A.J., Macphail, R.I. (Eds.), Archaeological Sediments and Soils: Analysis, Interpretation and Management. Institute of Archaeology, University College London, London, pp. 205–220.

Macklin, M.G., Dowsett, R.B., 1989. The chemical and physical speciation of trace metals in fine grained overbank flood sediments in the Tyne Basin, Northeast England. Catena 16, 135–151.

Macklin, M.G., Klimek, K., 1992. Dispersal, storage, and transformation of metal contaminated alluvium in the Upper Vistula Basin, southwest Poland. Applied Geography 12, 7–30.

Macklin, M.G., Lewin, J., 1989. Sediment transfer and transformation of an alluvial valley floor: The River South Tyne, Northumbria, U.K. Earth Surface Processes and Landforms 14, 233–246.

Macklin, M.G., Lewin, J., 2008. Alluvial responses to the changing Earth system. Earth Surface Processes and Landforms 33, 1374–1395.

Macklin, M.G., Lewin, J., 2019. River stresses in anthropogenic times: Large-scale global patterns and extended environmental timelines. Progress in Physical Geography 43 (1), 3–23.

Macklin, M.G., Ridgway, J., Passmore, D.G., Rumsby, B.T., 1994. The use of overbank sediment for geochemical mapping and contamination assessment: Results from selected English and Welsh floodplains. Applied Geochemistry 9 (6), 689–700.

Macklin, M.G., Hudson-Edwards, K.A., Dawson, E.J., 1997. The significance of pollution from historic metal mining in the Pennine orefields on river sediment contaminant fluxes to the North Sea. The Science of the Total Environment 194-195, 391–397.

Macklin, M.G., Brewer, P.A., Hudson-Edwards, K.A., et al., 2006. A geomorphological approach to the management of river contaminated by metal mining. Geomorphology 79, 423–447.

Macklin, M.G., Jones, A.F., Lewin, J., 2010. River response to rapid Holocene environmental change: Evidence and explanation in British catchments. Quaternary Science Reviews 29, 1555–1576. https://doi.org/10.1016/j.quascirev.2009.06.010.

Macklin, M.G., Lewin, J., Jones, A.F., 2014. Anthropogenic alluvium: An evidence-based meta-analysis for the UK Holocene. Anthropocene 6, 26–38.

Macphail, R.I., Courty, M.A., Gebhardt, A., 1990. Soil micromorphological evidence of early agriculture in north-west Europe. World Archaeology 22 (1), 53–69.

Madej, M.A., Sutherland, D.G., Lisle, T.E., Pryor, B., 2009. Channel responses to varying sediment input: A flume experiment modeled after Redwood Creek, California. Geomorphology 103, 507–519.

Magilligan, F.J., 1985. Historical floodplain sedimentation in the Galena River basin, Wisconsin and Illinois. Annals of the Association of American Geographers 75, 583–594.

Magilligan, F.J., Buraas, E.M., Renshaw, C.E., 2015. The efficacy of stream power and flow duration on geomorphic responses to catastrophic flooding. Geomorphology 228, 175–188.

Mahler, B.J., Van Metre, P.C., Bashara, T.J., et al., 2005. Parking lot sealcoat: An unrecognized source of urban polycyclic aromatic hydrocarbons. Environmental Science and Technology 39, 5560–5566.

Malhotra, K., Lamba, J., Shepherd, S., 2020. Sources of stream bed sediment in an urbanized watershed. Catena 184, 104228.

Mantei, E.J., Coonrod, D.D., 1989. Heavy metal content in the stream sediments adjacent to a sanitary landfill. Environmental Geology and Water Sciences 13, 51–58.

Marcus, W.A., 1987. Copper dispersion in ephemeral stream sediments. Earth Surface Processes and Landforms 12, 217–228.

Marcus, W.A., 1989. Dilution mixing estimates of trace metal concentrations in suspended sediments. Environmental Geology and Water Science 14 (3), 213–219.

Marcus, W.A., Meyer, G.A., Nimmo, D.R., 2001. Geomorphic control of persistent mine impacts in a Yellowstone Park stream and implications for the recovery of fluvial systems. Geology 29 (4), 355–358.

Marron, D.C., 1989. Physical and chemical characteristics of a metal-contaminated overbank deposit, west central South Dakota, U.S.A. Earth Surface Processes and Landforms 14, 419–432.

Marron, D.C., 1992. Floodplain storage of mine tailings in the Belle Fourche River system: A sediment budget approach. Earth Surface Processes and Landforms 17, 675–685.

Marsh, G.P., 1864. Man and Nature; Physical Geography as Modified by Human Action. Charles Scribner, New York, NY, 594 pp.

Martin, C.W., 2004. Heavy metal storage in near channel sediment of the Lahn River, Germany. Geomorphology 3-4, 275–285.

Martin, J.M., Meybeck, M., 1979. Elemental mass-balance of material carried by major world rivers. Marine Chemistry 7, 173–206.

Matys Grygar, T., Novkova, T., Babek, O., et al., 2013. Robust assessment of moderate heavy metal contamination levels in floodplain sediments: A case study on the Jizera River, Czech Republic. Science of the Total Environment 452-453, 233–245.

Maxfield, D., Rodriguez, J.M., Buettner, M., et al., 1970. Heavy metal content in the sediments of the southern part of the Coeur d'Alene Lake. Environmental Pollution 6 (4), 263–266.

McBride, M., Booth, D.B., 2005. Urban impacts on physical stream condition: Effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association 41 (3), 565–580.

McKean, J.A., Dietrich, W.E., Finkel, R.C., et al., 1993. Quantification of soil production and downslope creep rates from cosmogenic 10-Be accumulations on a hillslope profile. Geology 21, 343–346.

Meade, R.H., 1969. Errors in using modern stream-load data to estimate natural rates of denudation. Geological Society of America Bulletin 80, 1265–1274.

Melville, E.G.K., 1994. A Plague of Sheep: Environmental Consequences of the Conquest of Mexico. Cambridge University Press, Cambridge.

Meybeck, M., 1976. Total mineral dissolved transport by world major rivers. Hydrological Sciences Bulletin 21, 265–284.

Meyer, L.D., 1981. How rainfall intensity affects interrill erosion. Transactions of the American Society of Agricultural Engineers 24 (6), 1472–1475.

Micklin, P.P., 1988. Desiccation of the Aral Sea: A water management disaster in the Soviet Union. Science 241, 1170–1176.

Miller, J.R., 1997. The role of fluvial geomorphic processes in the dispersal of heavy metals from mine sites. Journal of Geochemical Exploration 58 (2–3), 101–118.

Miller, J.R., Orbock Miller, S.M., 2007. Contaminated rivers: A geomorphological–geochemical approach to site assessment and remediation. Springer, The Netherlands.

Miller, J.R., Rowland, J., Lechler, P.J., et al., 1996. Dispersal of mercury-contaminated sediments by geomorphic processes, Sixmile Canyon, Nevada, USA: Implication to site characterization and remediation of fluvial environments. Water, Air, and Soil Pollution 86, 373–388.

Milliman, J.D., Meade, R.H., 1983. World wide delivery of river sediment to the oceans. Journal of Geology 91, 1–21.

Milliman, J.D., Syvitski, J.P.M., 1992. Geomorphic/tectonic control of sediment discharge to the ocean: The importance of small mountainous rivers. Journal of Geology 100, 525–544.

Milliman, J.D., Qin, Y.-S., Ren, M.-E., Saito, Y., 1987. Man's influence on the erosion and transport of sediment by Asian rivers: The Yellow River (Huanghe) example. Journal of Geology 95, 751–762.

Moglen, G.E., 2000. Effect of orientation of spatially distributed curve numbers in runoff calculations. Journal of the American Water Resources Association 36 (6), 1391–1400.

Montgomery, D.R., 2008. Dreams of natural streams. Science 319, 291–292.

Montgomery, D.R., Buffington, J.M., 1997. Channel-reach morphology in mountain drainage basins. Geological Society of America Bulletin 109 (5), 596–611.

Montgomery, D.R., MacDonald, L.H., 2002. Diagnostic approach to stream channel assessment and monitoring. Journal of the American Water Resources Association 38 (1), 1–16.

Moore, A.M.T., 1982. Agricultural origins in the near East: A model for the 1980s. World Archaeology 14 (2), 224–236.

Moore, J.N., Langner, H.W., 2012. Can a river heal itself? Natural attenuation of metal contamination in river sediment. Environmental Science and Technology 46 (5), 2616–2623.

Moore, J.N., Luoma, S.N., 1990. Hazardous wastes from large-scale metal extraction: A case study. Environmental Science and Technology 24 (9), 1278–1285.

Moore, J.N., Brook, E.J., Johns, C., 1989. Grain size partitioning of metals in contaminated coarse-grained river floodplain river floodplain sediment: Clark Fork River, Montana, U.S.A. Environmental Geology and Water Science 14 (2), 107–115.

Moran, P.W., Noweell, L.H., Kemble, N.E., et al., 2017. Influence of sediment chemistry and sediment toxicity on macroinvertebrate communities across 99 wadeable streams of the Midwestern USA. Science of the Total Environment 599-600, 1469–1478.

Morgan, R.P.C., Quinton, J.N., Smith, R.E., et al., 1998. The European Soil Erosion Model (EUROSEM): A dynamic approach for predicting sediment transport from fields and small catchments. Earth Surface Processes and Landforms 23, 527–544.
Morin, E., Macaire, J.-J., Hinschberger, F., et al., 2011. Spatio-temporal evolution of the Choisille River (southern Parisian Basin, France) during the Weichselian and the Holocene as a record of climate trend and human activity in north-western Europe. Quaternary Science Reviews 30, 347–363. https://doi.org/10.1016/j.quascirev.2010.11.015.
Muchler, C.K., Young, R.A., 1975. Soil detachment by raindrops. In: Present and Prospective Technology for Predicting Sediment Yields and Sources. Proceedings of the Sediment-Yield Workshop. USDA Sediment Laboratory, Oxford, MS, 28–30 November 1972. ARS-S-40Agricultural Research Service, Washington, DC, pp. 113–117.
Munoz, S.E., Schroeder, S., Fike, D.A., Williams, J.W., 2014. A record of sustained prehistoric and historic land use from the Cahokia region, Illinois, USA. Geology 42, 499–502. https://doi.org/10.1130/G35541.1.
Murphy, G.E.P., Romanuk, T.N., 2014. A meta-analysis of declines in local species richness from human disturbances. Ecology and Evolution 4 (1), 91–103.
National Research Council (NRC), 1992. Water Transfers in the West: Efficiency, Equity, and the Environment. National Academy Press, Washington, DC, p. 320.
Nearing, M.A., Foster, G.R., Lane, L.J., Finkner, S.C., 1989. A process-based soil erosion model for USDA-Water Erosion Prediction Project technology. Transactions of the American Society of Agricultural Engineers 32, 1587–1593.
Neller, R.J., 1988. A comparison of channel erosion in small urban and rural catchments, Armidale New South Wales. Earth Surface Processes and Landforms 13, 1–7.
Neller, R.J., 1989. Induced channel enlargement in small urban catchments, Armidale, new South Wales. Environmental Geology and Water Sciences 14 (3), 167–171.
Nelson, A.D., Church, M., 2012. Placer mining along the Fraser River, British Columbia: The geomorphic impact. Geological Society of America Bulletin 124, 1212–1228.
Nevle, R.J., Bird, D.K., 2008. Effects of syn-pandemic fire reduction and reforestation in the tropical Americas on atmospheric CO_2 during European conquest. Palaeogeography, Palaeoclimatology, Palaeoecology 264, 25–38.
Newson, M.D., Large, A.R.G., 2006. 'Natural' rivers, 'hydromorphological quality' and river restoration: A challenging new agenda for applied fluvial geomorphology. Earth Surface Processes and Landforms 31 (13), 1606–1624.
Nicholas, A.P., Ashworth, P.J., Kirkby, M.J., et al., 1995. Sediment slugs: Large-scale fluctuations in fluvial sediment transport rates and storage volumes. Progress in Physical Geography 19 (4), 500–519.
Niezgoda, S.L., Johnson, P.A., 2005. Improving the urban stream restoration effort: Identifying critical form and processes and relationships. Environmental Management 35 (5), 579–592.
Notebaert, B., Verstraeten, G., 2010. Sensitivity of West and Central European river systems to environmental changes during the Holocene: A review. Earth-Science Reviews 103, 163–182. https://doi.org/10.1016/j.earscirev.2010.09.009.
Notebaert, B., Broothaerts, N., Verstraeten, G., 2018. Evidence of anthropogenic tipping points in fluvial dynamics in Europe. Global Planetary Change 164, 27–38. https://doi.org/10.1016/j.gloplacha.2018.02.008.
Novotny, V., Chesters, G., 1989. Delivery of sediment and pollutants from nonpoint sources: A water quality perspective. Journal Soil and Water Conservation 44 (6), 568–576.
O'Driscoll, M., Clinton, S., Jefferson, A., et al., 2010. Urbanization effects on watershed hydrology and in-stream processes in the southern United States. Water 2, 605–648.
O'Hara, S.L., Street-Perrott, F.A., Burt, T.P., 1993. Accelerated soil erosion around a Mexican highland lake caused by prehispanic agriculture. Nature 362, 48–51.
Olley, J.M., Wasson, R.J., 2003. Changes in the flux of sediment in the Upper Murrumbidgee catchment, Southeastern Australia, since European settlement. Hydrologic Processes 17, 3307–3320.
Olley, J.M., Murray, A.S., Mackenzie, D.H., Edwards, K., 1993. Identifying sediment sources in gullied catchments using natural and anthropogenic radioactivity. Water Resources Research 29 (4), 1037–1043.
Ongley, E.D., 1987. Scale effects in fluvial sediment-associated chemical data. Hydrological Processes 1 (2), 171–179.
Opperman, J.J., Galloway, G.E., Fargione, J., et al., 2009. Sustainable floodplains through large-scale reconnection to rivers. Science 326, 1487–1488.
Ottesen, R.T., Bogen, J., Bolviken, B., Volden, T., 1989. Overbank sediment: A representative sample medium for regional geochemical mapping. Journal of Geochemical Exploration 32 (1–3), 257–277.
Owen, M.R., Pavlowsky, R.T., Womble, P.J., 2011. Historical disturbance and contemporary floodplain development along and Ozark River, southwest Missouri. Physical Geography 32 (5), 423–444.
Owens, P.N., Walling, D.E., Carton, J., et al., 2001. Downstream changes in the transport and storage of sediment-associated contaminants (P, Cr, and PCBs) in agricultural and industrialized drainage basins. Science of the Total Environment 266 (1–3), 177–186.
Paine, A.D.M., 1985. Ergodic reasoning in geomorphology: Time for a review of the term? Progress in Physical Geography 9 (1), 1–15. https://doi.org/10.1177/030913338500900101.
Pandey, A., Himanshu, S.K., Mishra, S.K., Singh, V.P., 2016. Physically based soil erosion and sediment yield models revisited. Catena 147, 595–620.
Parsons, A.J., Wainwright, J., Brazier, R.E., Powell, D.M., 2006. Is sediment delivery a fallacy? Earth Surface Processes and Landforms 31, 1325–1328.
Pavlowsky, R.T., 1996. Fluvial transport and long-term mobility of mining-related zinc. In: Tailings and Mine Waste '96 Proceedings of the Third International Conference on Tailings and Mine Waste '96, Fort Collins, Colorado, USA, January 16–19, 1996, pp. 395–404. A.A. Balkema, Rotterdam, The Netherlands. ISBN 9054105941.
Pavlowsky, R.T., 2013. Coal-tar pavement sealant use and polycyclic aromatic hydrocarbon contamination in urban stream sediments. Physical Geography 34 (4–5), 392–415.
Pavlowsky, R.T., Lecce, S.A., Bassett, G., Martin, D.J., 2010. Legacy Hg-Cu contamination of active stream sediments in the Gold Hill Mining District, North Carolina. Southeastern Geographer 50 (4), 503–522.
Pavlowsky, R.T., Lecce, S.A., Owen, M.R., Martin, D.J., 2017. Legacy sediment, lead, and zinc storage in channel and floodplain deposits of the Big River, Old Lead Belt Mining District, Missouri, USA. Geomorphology 299, 54–75.
Pelletier, J.D., Murray, A.B., Pierce, J.L., et al., 2015. Forecasting the response of Earth's surface to future climatic and land use changes: A review of methods and research needs. Earth's Future 3, 220–251. https://doi.org/10.1002/2014EF000290.
Perhac, R.M., Whelan, C.J., 1972. A comparison of water, suspended soilid, and bottom sediment analyses for geochemical prospecting in a northeast Tennessee zinc district. Journal of Geochemical Exploration 1 (1), 47–53.
Phillips, J.D., 1988. Nonpoint source pollution and spatial aspects of risk assessment. Annual of the Association of American Geographers 78 (4), 611–623.
Phillips, J.D., Van Dyke, C., 2017. State-and-transition models in geomorphology. Catena 153, 168–181. https://doi.org/10.1016/j.catena.2017.02.009.
Piegay, H., Darby, S.E., Mosselman, E., Surian, N., 2005. A review of techniques available for delimiting the erodible river corridor: A sustainable approach to managing bank erosion. River Research and Applications 21, 773–789.
Pielke Sr., R.A., Pitman, A., Niyogi, D., et al., 2011. Land use/land cover changes and climate: Modeling analysis and observational evidence. Wiley Interdisciplinary Reviews, Climate Change 2 (6), 828–850. https://doi.org/10.1002/wcc.144.
Pimm, S.L., Raven, P., 2000. Extinction by numbers. Nature 403, 843–845.
Pitlick, J., 1997. A regional perspective of the hydrology of the 1993 Mississippi River basin floods. Annals of the Association of American Geographers 87, 135–151.
Pizzuto, J.E., Hession, W.C., McBride, M., 2000. Comparing gravel-bed rivers in paired urban and rural catchments of southeastern Pennsylvania. Geology 28 (1), 79–82.
Pizzuto, J., Skalak, K., Pearson, A., Benthem, A., 2016. Active overbank deposition during the last century, South River, Virginia. Geomorphology 257, 164–178.
Plass, G.N., 1956. Effect of carbon dioxide variations on climate. American Journal of Physics 24 (5), 376–387.
Playfair, J., 1802. Illustrations of the Huttonian Theory of the Earth. Cadell and Davies, London, and William Creech, Edinburgh, 528 pp.
Poesen, J., 2018. Soil erosion in the Anthropocene: Research needs. Earth Surface Processes and Landforms 43, 64–84.
Poff, N.L., Bledsoe, B.P., Cuhaciyan, C.O., 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology 79, 264–285.
Pohl, M.D., Pope, K.O., Jones, J.G., et al., 1996. Early agriculture in the Maya Lowlands. Latin American Antiquity 7 (4), 355–372.

Pongratz, J., Reick, C., Raddatz, T., Claussen, M., 2008. A reconstruction of global agricultural areas and land cover for the last millennium. Global Biogeochemical Cycles 22. https://doi.org/10.1029/2007GB003153. GB3018.
Pongratz, J., Reick, C., Raddatz, T., Claussen, M., 2009. Effects of anthropogenic land cover change on the carbon cycle of the last millennium. Global Biogeochemical Cycles 23. GB4001.
Portenga, E.W., Westwaway, K.E., Bishop, P., 2016. Timing of post-European settlement alluvium deposition in SE Australia: A legacy of European land-use in the Goulburn Plains. The Holocene 26 (9), 1472–1485.
Potter, K.W., 1991. Hydrological impacts of changing land management practices in a moderately sized agricultural catchment. Water Resources Research 27 (5), 845–855.
Presley, B.J., Trefry, J.H., Shokes, R.F., 1980. Heavy metal inputs to Mississippi Delta sediments: A historical view. Water, Air, and Soil Pollution 13, 481–494.
Pringle, H., 1998. The slow birth of agriculture. Science 282, 1446.
Quine, T.A., Walling, D.E., 1993. Use of ceasium-137 measurements to investigate relationships between erosion rates and topography. In: Thomas, D.S.G., Allison, R.J. (Eds.), Landscape Sensitivity. Wiley, New York, NY, pp. 29–48.
Rang, M.C., Schouten, C.J., 1989. Evidence for historical metal pollution in floodplain soils: The Meuse. In: Petts, G.E. (Ed.), Historical Change of Large Alluvial Rivers: Western Europe. John Wiley and Sons Ltd., pp. 127–142
Reece, D.E., Felkey, J.R., Wai, C.M., 1978. Heavy metal pollution in the sediment of the Coeur d'Alene River, Idaho. Environmental Geology 2, 289–293.
Reid, H.E., Brierley, G.J., 2015. Assessing geomorphic sensitivity in relation to river capacity for adjustment. Geomorphology 251, 108–121.
Reid, H.E., Gregory, C.E., Brierly, G.J., 2008. Measures of physical heterogeneity in appraisal of geomorphic river condition for urban streams: Twin Streams Catchment, Auckland, New Zealand. Physical Geography 29 (3), 247–274.
Reilly, J., Maggio, P., Karp, S., 2004. A model to predict impervious surface for regional and municipal land use planning purposes. Environmental Impact Assessment Review 24 (3), 363–382.
Reneau, S.L., Drakos, P.G., Katxman, D., et al., 2004. Geomorphic controls on contaminant distribution along an ephemeral stream. Earth Surface Processes and Landforms 29, 1209–1223.
Renfro, G.W., 1975. Use of erosion equations and sediment-delivery ratios for predicting sediment yield. In: Proceedings of the Sediment-Yield Workshop. USDA Sediment Laboratory, Oxford, MS, 28–30 November 1972. ARS-S-40Agricultural Research Service, Wash., D.C., pp. 33–45
Renwick, W.H., Smith, S.V., Bartley, J.D., Buddemeier, R.W., 2005. The role of impoundments in the sediment budget of the conterminous United States. Geomorphology 71, 99–111.
Renwick, W.H., Sleezer, R.O., Buddemeier, R.W., Smith, S.V., 2006. Small artificial ponds in the United States: Impacts on sedimentation and carbon budget. In: Proceedings of the 8th Federal Interagency Sedimentation Conference Paper 10A-3, 7 pp.
Ress, L.D., Hung, C.-L., James, L.A., 2020. Impacts of urban drainage systems on stormwater hydrology: Rocky Branch watershed, Columbia, South Carolina. Journal of Flood Risk Management 13 (3). https://doi.org/10.1111/jfr3.12643.
Rhoads, B.L., Cahill, R.A., 1999. Geomorphological assessment of sediment contamination in an urban stream system. Applied Geochemistry 14 (4), 459–483.
Ritchie, J.C., Spraberry, J.A., McHenry, J.R., 1974. Estimating soil loss from the redistribution of fallout 137Cs. Soil Science Society of America Proceedings 38, 137–139.
Rittenhouse, G., 1943. Transportation and deposition of heavy minerals. Bulletin of the Geological Society of America 54, 1725–1780.
Roehl, J.W., 1962. Sediment source areas, delivery ratios and influencing morphological factors. International Association of Hydrological Sciences 59, 202–213.
Rogger, M., Agnoletti, M., Alaoui, A., et al., 2017. Land use change impacts on floods at the catchment scale: Challenges and opportunities for future research. Water Resources Research 53, 5209–5219. https://doi.org/10.1002/2017WR020723.
Romano, M., 2015. Reviewing the term uniformitarianism in modern Earth sciences. Earth-Science Reviews 148, 65–76.
Rose, A.W., Dahlberg, E.C., Keith, M.L., 1970. A multiple regression technique for adjusting background values in stream sediment geochemistry. Economic Geology 65, 156–165.
Rosenmeier, M.F., Hodell, D.A., Brenner, M., et al., 2002. A 4,000-year lacustrine record of environmental change in the Southern Maya Lowlands, Peten, Guatemala. Quaternary Research 57 (2), 183–190.
Rowan, J.S., Barnes, S.J.A., Hetherington, S.L., et al., 1995. Geomorphology and pollution: The environmental impacts of lead mining, Leadhills, Scotland. Journal of Geochemical Exploration 52 (1–2), 57–65.
Royall, D., Kennedy, L., 2016. Historical erosion and sedimentation in two small watersheds of the southern Blue Ridge Mountains, North Carolina, USA. Catena 143, 174–186.
Rubey, W.W., 1933. The size-distribution of heavy minerals within a water-laid sandstone. Journal of Sedimentary Petrology 3, 3–29.
Ruddiman, W.F., 2003. The anthropogenic greenhouse era began thousands of years ago. Climatic Change 61, 261–293.
Ruddiman, W.F., 2007. The early anthropogenic hypothesis: Challenges and responses. Reviews of Geophysics 45. RG000207R.
Ruddiman, W.F., Ellis, E.C., 2009. Effect of per-capita land use changes on Holocene forest clearance and CO_2 emissions. Quaternary Science Reviews 28 (27–28), 3011–3015.
Ruddiman, W.F., Ellis, E.C., Kaplan, J.O., Fuller, D.Q., 2015. Defining the epoch we live in. Is a formally designated 'Anthropocene' a good idea? Science 348 (6130), 38–39.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2017. Global sediment yields from urban and urbanizing watersheds. Earth-Science Reviews 168, 73–80.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2018. Urban catchment runoff increases bedload sediment yield and particle size in channels. Anthropocene 23, 53–66.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2019a. A suburban sediment budget: Coarse-grained sediment flux through hillslopes, stormwater systems, and streams. Earth Surf. Process. Landforms 44, 2600–2614.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2019b. Urban sediment supply to the streams from hillslope sources. Science of the Total Environment 653, 684–697.
Russell, K.L., Vietz, G.J., Fletcher, T.D., 2020. How urban streamwater regimes drive geomorphic degradation of receiving streams. Progress in Physical Geography: Earth and Environment. https://doi.org/10.1177/0309133319893927.
Sala, O.E., Chapin III, F.S., Armesto, J.J., et al., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770–1774.
Salomons, W., 1985. Sediments and water quality. Environmental Technology Letters 6 (1–11), 315–326.
Salomons, W., 1995. Environmental impact of metals derived from mining activities: Processes, predictions, prevention. Journal of Geochemical Exploration 52, 5–23.
Saxton, N.E., Olley, J.M., Smith, S., Ward, D.P., Rose, C.W., 2012. Gully erosion in sub-tropical south-east Queensland, Australia. Geomorphology 173-174, 80–87.
Sayre, W.W., Hubbell, D.W., 1965. Transport and dispersion of labeled bed material, North Loop River, Nebraska. In: United States Geological Survey Professional Paper 433-C. Government Printing Office, Washington D.C.
Sayre, W.W., Guy, H.P., Chamberlain, A.R., 1963. Uptake and transport of radionuclides by stream sediments. USGS Professional Paper 433-A. U.S. Government Printing Office, Washington D.C.
Schillereff, D.N., Chiverrell, R.C., Macdonald, N., et al., 2016. Quantifying system disturbance and recovery from historical mining-derived metals contamination at Brotherswater, northwest England. Journal of Paleolimnology 56, 205–221.
Schumm, S.A., 1977. The Fluvial System. Wiley, New York, NY.
Schumm, S.A., 1979. Geomorphic thresholds, the concept and its applications. Transactions of the Institute of British Geographers, New Series 4 (4), 485–515.
Schumm, S.A., 1991. To Interpret the Earth: Ten Ways to be Wrong. Cambridge University Press, New York, NY, p. 131.
Schumm, S.A., 2005. River Variability and Complexity. Cambridge University Press, Cambridge, p. 220.
Schumm, S.A., Lichty, R.W., 1965. Time, space and causality in geomorphology. American Journal of Science 263, 110–119.
Schumm, S.A., Harvey, M.D., Watson, C.C., 1984. Incised Channels: Dynamics, Morphology and Control. Water Resources Publications, Littleton, CO, 200 pp.
Sear, D., Carver, S., 1996. The release and dispersal of Pb and Zn contaminated sediments within an Arctic braided river system. Applied Geochemistry 11 (1–2), 187–195.
Shacklette, H.T., Boerngen, J.G., 1984. Element Concentrations in Soils and Other Surficial Materials of the Conterminous United States. In: U.S. Geological Survey Professional Paper 1270, 105 pp.

Shahrestani, S., Mokhtari, A.R., 2017. Improved detection of anomalous catchment basins by incorporating drainage density in dilution correction of geochemical results. Geochemistry: Exploration, Environment, Analysis 17, 194–203.

Shellberg, J.G., Spencer, J., Brooks, A.P., et al., 2016. Degradation of the Mitchell River fluvial megafan by alluvial gully erosion increased by post-European land use change, Queensland, Australia. Geomorphology 266, 105–120. https://doi.org/10.1016/j.geomorph.2016.04.021.

Shepherd, S.L., Dixon, J.C., Davis, R.K., Feinstein, R., 2010. The effect of land use on channel geometry and sediment distribution in gravel mantled bedrock stream, Illinois River watershed, Arkansas. River Research and Applications 27, 857–866.

Shi, Z.H., Huang, X.D., Ai, L., et al., 2014. Quantitative analysis of factors controlling sediment yield in mountainous watersheds. Geomorphology 226, 193–201.

Simon, A., 1989. A model of channel response in disturbed alluvial channels. Earth Surface Processes and Landforms 14, 11–26.

Simon, A., Hupp, C.R., 1986. Channel evolution in modified Tennessee channels. In: Proceedings of the Fourth Federal Interagency Sedimentation Conference. Las Vegas, Nevada, 24–27 March 1986, vol. 2. Government Printing Office, Washington, DC, 5-71–5-82.

Simon, A., Rinaldi, M., 2006. Disturbance, stream incision and channel evolution: The roles of excess transport capacity and boundary materials in controlling channel response. Geomorphology 79, 361–383.

Simons, R.K., Simons, D.B., 1993. An analysis of Platte River channel changes. In: Schumm, S.A., Winkley, B.R. (Eds.), The Variability of Large Alluvial Rivers. American Society of Civil Engineers Press, New York, NY, pp. 341–361.

Sims, A.J., Rutherfurd, I.D., 2017. Management responses to pulses of bedload sediment in rivers. Geomorphology 70–86. https://doi.org/10.1016/j.geomorph.2017.04.010.

Singer, M.B., Aalto, R., James, L.A., et al., 2013. Enduring legacy of a toxic fan via episodic redistribution of California gold mining debris. Proceedings of the National Academy of Sciences of the United States of America 110 (46), 18436–18441. https://doi.org/10.1073/pnas.1302295110.

Sklar, L.S., Fadde, J., Venditti, J.G., Nelson, P., Wydzga, M.A., Cui, Y., Dietrich, W.E., 2009. Translation and dispersion of sediment pulses in flume experiments simulating gravel augmentation below dams. Water Resource Research 45. https://doi.org/10.1029/2008WR007346.

Skoglund, P., Malmstrom, H., Raghavan, M., et al., 2012. Origins and genetic legacy of Neolithic farmers and hunter-gatherers in Europe. Science 336 (6080), 466–469. https://doi.org/10.1126/science.1216304.

Slaymaker, O., Spencer, T., Dadson, S., 2009. Landscape and landscape-scale processes as the unfilled niche in the global environmental change debate, an introduction. In: Slaymaker, O., Spencer, T., Embleton-Hamann, C. (Eds.), Geomorphology and Global Environmental Change. Cambridge University Press, Cambridge, pp. 1–36.

Small, E., Anderson, R., Hancock, G., 1999. Estimates of the rate of regolith production using ^{10}Be and ^{26}Al from an Alpine hillslope. Geomorphology 27, 131–150.

Smith, B.D., 1998. Between foraging and farming. Science 270 (5357), 1651.

Smith, S.V., Renwick, W.H., Bartley, J.D., Buddemeier, R.W., 2002. Distribution and significance of small, artificial water bodies across the United States landscape. Science of the Total Environment 299, 21–36.

Stankowski, S.J., 1972. Population density as an indirect indicator of urban and suburban land-surface modifications. In: US Geological Survey Professional Paper 800-B.

Steele, K.F., Wagner, G.H., 1975. Trace metal relationships in bottom sediments of a fresh water stream—the Buffalo River, Arkansas. Journal of Sedimentary Petrology 45 (1), 310–319.

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O., Ludwig, C., 2015. The trajectory of the Anthropocene: The Great Acceleration. The Anthropocene Review 2 (1), 81–98. https://doi.org/10.1177/2053019614564785.

Stinchcomb, G.E., Messner, T.C., Driese, S.G., et al., 2011. Precolonial (AD 1100-1600) sedimentation related to prehistoric maize agriculture and climate change in eastern North America. Geology 39, 363–366.

Stinchcomb, G.E., Stewart, R.M., Messner, T.C., et al., 2013. Using event stratigraphy to map the Anthropocene: An example from the historic coal mining region in eastern Pennsylvania, USA. Anthropocene 2, 42–50.

Stoffel, M., Wyzga, B., Marston, R.A., 2016. Floods in mountain environments: A synthesis. Geomorphology 272, 1–9.

Stone Jr., B., 2004. Paving over paradise: How land use regulations promote residential imperviousness. Landscape and Urban Planning 69, 101–113.

Stutenbecker, L., Costa, A., Bakker, M., Anghileri, D., Molnar, P., Lane, S.N., Schlunegger, F., 2019. Disentangling human impact from natural controls of sediment dynamics in an Alpine catchment. Earth Surface Processes and Landforms 44, 2885–2902. https://doi.org/10.1002/esp.4716.

Surian, N., Righini, M., Lucía, A., et al., 2016. Channel response to extreme floods: Insights on controlling factors from six mountain rivers in northern Apennines, Italy. Geomorphology 272, 78–91.

Sutherland, R.A., 2000. Bed sediment-associated trace metals in an urban stream, Oahu, Hawaii. Environmental Geology 39, 611–627.

Sutherland, R.A., Tack, F.M.G., Tolosa, C.A., Verloo, M.G., 2000. Operationally defined metal fractions in road deposited sediment, Honolulu, Hawaii. Journal of Environmental Quality 29 (5), 1431–1439.

Swennen, R., Van Keer, I., De Vos, W., 1994. Heavy metal contamination in overbank sediments of the Geul River (East Belgium): Its relation to former Pb-Zn mining activities. Environmental Geology 24, 12–21.

Symader, W., 1980. Suspended heavy metals: An investigation of their temporal behavior in flowing waters. Catena 7, 1–26.

Symader, W., Thomas, W., 1978. Interpretation of heavy metal pollution in flowing waters and sediment by means of hierarchical grouping analysis using two different error indices. Catena 5, 131–144.

Syvitski, J.P.M., 2003. Supply and flux of sediment along hydrological pathways: Research for the 21st century. Global and Planetary Change 39, 1–11.

Syvitski, J.P.M., Vörösmarty, C.J., Kettner, A.J., Green, P., 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. Science 308, 376–380.

Szabo, Z., Buro, B., Szabo, J., et al., 2020. Geomorphology as a driver of heavy metal accumulation patterns in a floodplain. Water 12, 563. https://doi.org/10.3390/w12020563.

Taggart, A.F., 1945. Handbook of Mineral Dressing: Ores and Industrial Minerals. John Wiley and Sons, New York.

Taniguchi, K.T., Biggs, T.W., Langendoen, E.J., et al., 2018. Stream channel erosion in a rapidly urbanizing region of the US-Mexico border: Documenting the important of channel hardpoints with structure-from-Motion photogrammetry, 2018. Earth Surface Processes and Landforms 43 (7), 1465–1477.

Taylor, S.R., 1964. Abundance of chemical elements in the continental crust—A new table. Geochemica Cosmochimica et Acta 28, 1273–1285.

Taylor, M.P., 1996. The variability of heavy metals in floodplain sediments: A case study from mid Wales. Catena 28 (1–2), 71–87.

Taylor, M.P., 2007. Distribution and storage of sediment-associated heavy metals downstream of the remediated Rum Jungle Mine on the East Branch of the Finniss River, Northern Territory, Australia. Journal of Geochemical Exploration 92 (1), 55–72.

Taylor, M.P., Little, J.A., 2013. Environmental impact of a major copper mine spill on a river and floodplain system. Anthropocene 3, 36–50.

Taylor, K.G., Owens, P.N., 2009. Sediments in urban river basins: A review of sediment-contaminant dynamics in an environmental system conditioned by human activities. Journal of Soils and Sediments 9, 281–303.

Thevenot, R.R., Moilleron, R., Letel, L., et al., 2007. Critical budget of metal sources and pathways in the Seine River basin (1994–2003) for Cd, Cr, Cu, Hg, Ni, Pb and Zn. Science of the Total Environment 375, 180–203.

Thomas, M.F., 2001. Landscape sensitivity in time and space – An introduction. Catena 42, 83–98.

Thomas Jr., W.L. (Ed.), 1956. Man's Role in Changing the Face of the Earth. University of Chicago Press, Chicago, p. 1193.

Thompson, C., Croke, J., 2013. Geomorphic effects, flood power, and channel competence of a catastrophic flood in confined and unconfined reaches of the upper Lockyer valley, southeast Queensland, Australia. Geomorphology 197, 156–169.

Thompson, C.J., Croke, J., Fryirs, K., Grove, J.R., 2016. A channel evolution model for subtropical macrochannel systems. Catena 139, 199–213.

Thoms, M., Thiel, P., 1995. The impact of urbanization on the bed sediments of South Creek, New South Wales. Australian Geographic Studies 33 (1), 31–43.

Thorn, C.E., Welford, M.R., 1994. The equilibrium concept in geomorphology. Annals of the Association of American Geographers 84, 666–696.

Thornes, J.B., Brunsden, D., 1977. Geomorphology and Time. Wiley, New York, NY, p. 208.

Thornthwaite, C.W., 1956. Modification of rural microclimates. In: Thomas Jr., W.L. (Ed.), Man's Role in Changing the Face of the Earth. University of Chicago Press, Chicago, pp. 567–583.
Tollan, A., 2002. Land-use change and floods: What do we need most, research or management? Water Science and Technology 45 (8), 183–190.
Tourtelot, H.A., 1968. Hydraulic equivalence of grain of quartz and heavier minerals, and implications for the study of placers. In: Geological Survey Professional Paper 594-F.
Trimble, S.W., 1974. Man-Induced Soil Erosion on the Southern Piedmont, 1700–1970. Soil and Water Conservation Society of America, Ankeny, IA.
Trimble, S.W., 1977. The fallacy of stream equilibrium in contemporary denudation studies. American Journal of Science 277, 876–887.
Trimble, S.W., 1997. Contribution of stream channel erosion to sediment yield from an urbanizing watershed. Science 278, 1442–1444.
Troeh, F.R., Hobbs, J.A., Donahue, R.L., 2003. Soil and Water Conservation. Prentice Hall, Upper Saddle River, NJ, 656 pp.
Tsytsenko, A.V., 2003. The problems of the Aral Sea Basin. In: Shiklomanov, I.A., Rodda, J.C. (Eds.), World Water Resources at the Beginning of the Twenty-First Century. Cambridge University Press, Cambridge, pp. 145–156.
Turekian, K.K., Scott, M.R., 1967. Concentrations of Cr, Ag. Mo, Ni, Co, and Mn in suspended material in streams. Environmental Science and Technology 1 (11), 940–942.
Turekian, K.K., Wedepohl, K.H., 1961. Distribution of the elements of some major units of the earth's crust. Geological Society of American Bulletin 72, 175–192.
Turner II, B.L., Matson, P.A., McCarthy, J., et al., 2003. Illustrating the coupled human–environment system for vulnerability analysis: Three case studies. Proceedings of the National Academy of Sciences 100 (14), 8080–8085.
Turner II, B.L., Lambin, E.F., Reenberg, A., 2007. The emergence of land change science for global environmental change sustainability. PNAS 104 (52), 20666–20671. https://doi.org/10.1073/pnas.0704119104.
Turner, B.L.I.I., Clark, W.C., Kates, R.W., et al. (Eds.), 1990. The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere over the Past 300 Years. Cambridge University Press, Cambridge, p. 713.
Turner, J.N., Brewer, P.A., Macklin, M.G., 2008. Fluvial-controlled metal and As mobilization, dispersal, and storage in the Rio Guadiamar, SW Spain and its implications for long-term contaminant fluxes to the Donana wetlands. Science of the Total Environment 394 (1), 144–161.
US Department of Agriculture (USDA) and Agricultural Research Service (ARS), 2013. Revised Universal Soil Loss Equation Version 2 (RUSLE2). Wash., DC. https://www.ars.usda.gov/ARSUserFiles/60600505/RUSLE/RUSLE2_.
USEPA (U.S. Environmental Protection Agency), 2000. Low Impact Development (LID): A Literature Review. EPA-841-B-00-005. USEPA Office of Water, Washington, DC, 41 pp.
Van Andel, T.H., Runnels, C.N., Pope, K.O., 1986. Five thousand years of land use and abuse in the Southern Argolid, Greece. Hesperia 55, 103–128.
Van Andel, T.H., Zangger, E., Demitrack, A., 1990. Land use and soil erosion in prehistoric and historical Greece. Journal of Field Archaeology 17 (4), 379–396.
Van Dyke, C., 2013. Channels in the making–an appraisal of channel evolution models. Geography Compass 7 (11), 759–777.
Van Haveren, B.P., 1991. Placer mining and sediment problems in interior Alaska. In: Fan, S.S., Kuo, Y.H. (Eds.)Proceedings of the 5th Federal Interagency Sedimentation Conference. Government Printing Office, Washington, DC, 10-69–10-74.
Van Metre, P.C., Mahler, B.J., 2004. Contaminant trends in reservoir sediment cores as records of influent stream quality. Environmental Science and Technology 38, 2978–2986.
Vandenberghe, J., Vanacker, V., 2008. Towards a system approach in the study of river catchments. Geomorphology 98, 173–175.
Vanmaercke, M., Poesen, J., Govers, G., Verstraeten, G., 2015. Quantifying human impacts on catchment sediment yield: A continental approach. Global and Planetary Change 130, 22–36.
Vanoni, V.A. (Ed.), 1975. Sedimentation engineering. Manuals and Reports on Engineering Practice No. 54. ASCE, New York.
Vanwalleghem, T., Bork, H.R., Poesen, J., et al., 2006. Prehistoric and Roman gullying in the European loess belt: A case study from central Belgium. Holocene 16 (3), 393–401.
Vauclin, S., Mourier, B., Piégay, H., Winiarski, T., 2020. Legacy sediments in a European context: The example of infrastructure-induced sediments on the Rhône River. Anthropocene 100248.
Verstraeten, G., Prosser, I.P., 2008. Modelling the impact of land-use change and farm dam construction on hillslope sediment delivery to rivers at the regional scale. Geomorphology 98, 199–212.
Verstraeten, G., Broothaerts, N., Van Loo, M., et al., 2017. Variability in fluvial geomorphic response to anthropogenic disturbance. Geomorphology 294, 20–39.
Vietz, G.J., Hawley, R.J., 2019. Chapter 12- Protecting and managing stream morphology in urban catchments using WSUD. In: Sharma, A.K., Begbie, D., Gardner, T. (Eds.), Approaches to Water Sensitive Urban Design: Potential, Design, Ecological Health, Urban Greening, Economics, Policies, and Community Perceptions. Elsevier, pp. 249–267.
Vietz, G.J., Sammonds, M.J., Walsh, C.J., et al., 2014. Ecologically relevant geomorphic attributes of streams are impaired by even low levels of watershed effective imperviousness. Geomorphology 206, 67–78.
Vietz, G.J., Rutherford, I.D., Fletcher, T.D., Walsh, C.J., 2016a. Thinking out the channel: Challenges and opportunities for protection and restoration of stream morphology in urbanizing catchments. Landscape and Urban Planning 145, 34–44.
Vietz, G.J., Walsh, C.J., Fletcher, T.D., 2016b. Urban hydrogeomorphology and urban stream syndrome: Treating the symptoms and causes of geomorphic change. Progress in Physical Geography 40 (3), 480–492.
Vigiak, O., Malagó, A., Bouraoui, F., et al., 2015. Adapting SWAT hillslope erosion model to predict sediment concentrations and yields in large basins. Science of the Total Environment 538, 855–875.
Violin, C.R., Cada, P., Sudduth, E.B., et al., 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. Ecological Applications 21 (6), 1932–1949.
Vörösmarty, C.J., Green, P., Salisbury, J., Lammers, R., 2000. Global water resources: Vulnerability from climate change and population growth. Science 289 (5477), 284–288.
Wall, G.J., Wilding, L.P., Smeck, N.E., 1978. Physical, chemical, and mineralogical properties of fluvial uncondolidated bottom sediments in Northwestern Ohio. Journal of Environmental Quality 7 (3), 171–177.
Wallbrink, P.J., Murray, A.S., 1993. Use of fallout radionuclides as indicators of erosion processes. Hydrological Processes 7, 297–304.
Walling, D.E., 1983. The sediment delivery problem. Journal Hydrology 65, 209–237.
Walling, D.E., 1990. Linking the field to the river: Sediment delivery from agricultural land. In: Boardman, J., Foster, I.D.L., Dearing, J.A. (Eds.), Soil Erosion on Agricultural Land. Wiley, Chichester, pp. 129–152.
Walling, D.E., 2006. Human impact on land–ocean sediment transfers by the world's rivers. Geomorphology 79, 192–216.
Walling, D.E., Webb, B.W., 1983. Patterns of sediment yield. In: Gregory, K.J. (Ed.), Background to Palaeohydrology. Wiley, Chichester, UK, pp. 69–100.
Walling, D.E., Woodward, J.C., 1992. Use of radiometric fingerprints to derive information on suspended sediment sources. In: Bogen, J., Walling, D.E., Day, T.J. (Eds.), Erosion and Sediment Transport Monitoring Programmes in River Basins. IAHS Press, Wallingford, pp. 153–164. . IAHS Publication No. 210.
Walling, D.E., Owens, P.N., Carter, J., et al., 2003a. Storage of sediment-associated nutrients and contaminants in river channel and floodplain systems. Applied Geochemistry 18, 195–220.
Walling, D.E., Owens, P.N., Foster, I.D.L., Lees, J.A., 2003b. Changes in the fine sediment dynamics of the Ouse and Tweed basins in the UK over the last 100–150 years. Hydrologic Processes 17, 3245–3269.
Wallinga, J., 2002. Optically stimulated luminescence dating of fluvial deposits: A review. Boreas 31, 303–322.
Walsh, C.J., Roy, A.H., Feminella, J.W., et al., 2005. The urban stream syndrome: Current knowledge and the search for a cure. Journal of the North American Benthological Society 24 (3), 706–723.
Walter, R.C., Merritts, D.J., 2008. Natural streams and the legacy of water powered mills. Science 319 (5861), 299–304. https://doi.org/10.1126/science.1151716.
Wang, L., Leigh, D.S., 2015. Anthropic signatures in alluvium of the Upper Little Tennessee River valley, Southern Blue Ridge Mountains, USA. Anthropocene 11, 35–47.
Waslenchuk, D.G., 1975. Mercury in fluvial bed sediments subsequent to contamination. Environmental Geology 1, 131–136.

Waters, C.N., Zalasiewicz, J., Summerhayes, C., et al., 2016. The Anthropocene is functionally and stratigraphically distinct from the Holocene. Science 351. https://doi.org/10.1126/science.aad2622.

Wathen, S.J., Hoey, T.B., 1998. Morphological controls on the downstream passage of a sediment wave in a gravel-bed stream. Earth Surface Processes and Landforms 23, 715–730.

Wenger, S.J., Roy, A.H., Jackson, C.R., et al., 2009. Twenty-six key research questions in urban stream ecology: An assessment of the state of the science. Journal of the North American Benthological Society 28 (4), 1080–1098.

White, C.S., Gosz, J.R., 1983. Sediment chemistry as influenced by vegetation and bedrock in the southwestern U.S. Journal of the Water Resources Association 19 (5), 829–835.

Whitmore, T.M., Turner II, B.L., 2002. Cultivated Landscapes of Middle America on the Eve of Conquest. Oxford University Press, New York, NY.

Williams, G.P., Wolman, M.G., 1984. Downstream effects of dams on alluvial rivers. In: US Geological Survey Professional Paper 1286.

Wilson, C.G., Kuhnle, R.A., Bosch, D.D., et al., 2008. Quantifying relative contributions from sediment sources in Conservation Effects Assessment Project watersheds. Journal of Soil and Water Conservation 63 (6), 523–532.

Wischmeier, W.H., Smith, D.D., 1965. Predicting rainfall-erosion losses from cropland east of the Rocky Mountains. U.S. Department of Agriculture, Agricultural Research Service Publication S-40. US Government Printing Office, Washington, DC.

Wischmeier, W.H., Smith, D.D., 1978. Predicting rainfall-erosion losses: A guide to conservation planning. U.S. Department of Agriculture. In: Agricultural Handbook, p. 282.

Witter, B., Winkler, M., Friese, K., 2003. Depth distribution of chlorinated and polycyclic aromatic hydrocarbons in floodplain soils of the River Elbe. Acta Hydrochimica Et Hydrobiologica 4-5, 411–422.

Witter, A.E., Nguyen, M.H., Baidar, S., Sak, P.B., 2014. Coal-tar based sealcoated pavement: A major PAH source to urban stream sediments. Environmental Pollution 185, 59–68.

Wohl, E.E., 2000. Anthropogenic impacts on flood hazards. In: Wohl, E.E. (Ed.), Inland Flood Hazards Human, Riparian, and Aquatic Communities. Cambridge University Press, Cambridge, pp. 104–141.

Wohl, E.E., 2001. Virtual Rivers: Lessons from the Mountain Rivers of the Colorado Front Range. Yale University Press, New Haven, CT.

Wohl, E.E., 2015. Legacy effects on sediments in river corridors. Earth-Science Reviews 147, 30–53. https://doi.org/10.1016/j.earscirev.2015.05.001.

Wohl, E.E., Merritts, D.J., 2007. What is a natural river? Geography Compass 1 (4), 871–900.

Wohl, E.E., Lane, S.N., Wilcox, A.C., 2015. The science and practice of river restoration. Water Resources Research 51, 5974–5997. https://doi.org/10.1002/2014WR016874.

Wohl, E., Bledsoe, B.P., Jacobson, R.B., et al., 2015b. The natural sediment regime in rivers: Broadening the foundation for ecosystem management. Bioscience 65, 358–371.

Wolfenden, P.J., Lewin, J., 1977. Distribution of metal pollutants in floodplain sediments. Catena 4, 309–317.

Wolfenden, P.J., Lewin, J., 1978. Distribution of metal pollutants in active stream sediments. Catena 5, 67–78.

Wolman, M.G., 1967. A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler 49A, 385–395.

Wolman, M.G., 1977. Changing needs and opportunities in the sediment field. Water Resources Research 13 (1), 50–54.

Wolman, M.G., Miller, W.P., 1960. Magnitude and frequency of forces in geomorphic processes. Journal of Geology 68, 54–74.

Wolman, M.G., Schick, A.P., 1967. Effects of construction on fluvial sediment, urban and suburban areas of Maryland. Water Resources Research 3 (2), 451–464.

Woltemade, C.J., 1994. Form and process: Fluvial geomorphology and flood–flow interaction, Grant River, Wisconsin. Annals of the Association of American Geographers 84, 462–479.

Worrall, F., Burt, T.P., Howden, N.J.K., Hancock, G.R., 2014. Variation in suspended sediment yield across the UK - A failure of the concept and interpretation of the sediment delivery ratio. Journal of Hydrology 519, 1985–1996.

Wu, C.S., Murray, A.T., 2003. Estimating impervious surface distribution by spectral mixture analysis. Remote Sensing of Environment 84 (4), 493–505.

Wyzga, B., Ciszewski, D., 2010. Hydraulic control on the entrapment of heavy metal-polluted sediment on a floodplain of variable width, the upper Vistula River, southern Poland. Geomorphology 117, 272–286.

Xu, J., 2003. Sedimentation rates in the lower Yellow River over the past 2300 years as influenced by human activities and climate change. Hydrologic Processes 17, 3359–3371.

Yim, W.W.-S., 1981. Geochemical investigations on fluvial sediments contaminated by tin mine tailings, Cornwall, England. Environmental Geology 3, 245–256.

Zalasiewicz, J., Waters, C.N., Williams, M., et al., 2015. When did the Anthropocene begin? A mid-twentieth century boundary level is stratigraphically optimal. Quaternary International 383, 204–207. https://doi.org/10.1016/j.quaint.2014.11.045.

Zalasiewicz, J., Waters, C.N., Williams, M., Summerhayes, C.P. (Eds.), 2019. The Anthropocene as a Geological Time Unit: A Guide to the Scientific Evidence and Current Debate. Cambridge University Press, UK.

Zhang, X.C., 2019. Determining and modeling dominant processes of interrill soil erosion. Water Resources Research 55, 4–20. https://doi.org/10.1029/2018WR023217.