CHAPTER 5

Fine particles in small steepland streams: physical, ecological, and human connections

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Abstract

Fine particle dynamics in small steepland streams can be severely impacted by human activities, with ensuing effects on ecological and physical processes. Four components of fine particle dynamics are reviewed: sources and supply mechanisms; instream transport and deposition; biological impacts; and spatial and temporal scales of study and variability. Included within each topic is information on inorganic and organic particles, measurement and modelling techniques, and the impacts of human activities. Lastly, several remaining research needs are identified.

1 Introduction

Fine particulate matter can be an important component of many physical and biological processes in streams. In particular, deposition of fine sediment (<2 mm) has been repeatedly shown to degrade benthic habitat for fish and other organisms [1–3] and reduce water quality [4]. Fine-particulate organic matter (FPOM; 50–100 µm) can be an important flux linking up- and downstream reaches [5, 6] and can supply a significant amount of carbon to benthic invertebrates [7]. Furthermore, FPOM may play a large role in the movement and deposition of sorbed contaminants and nutrients [8]. However, both fine sediment and FPOM dynamics are impacted by human activities, such as logging and flow regulation. For example, logging can alter the dynamics of organic matter delivery and channel storage of fine particles [9–11].

Fine sediment and FPOM are often studied independently, despite the fact that suspended particles in many streams are composites of mineral and organic matter [12–14]. Interest in these composite particles (also known as aggregates or
flocs) has increased over the past decade, but most of the existing literature separates suspended load into inorganic and organic components, or considers all fine particles generally as “sediment”, usually assumed to be inorganic. In part, this one-sided focus may be due to disciplinary biases, but may also be logical for a given study, or necessary due to a limited knowledge of particle composition. In this review, we summarize information on fine sediment and organic particles as they are referred to in the literature – either as generic sediment or as independent components of the total suspended load. Although our review is somewhat limited in this regard, future study will benefit from attempts to link the dynamics of inorganic, organic, and aggregate particles. Progress in this field depends upon improved understanding of the interactions, differences, and similarities of these particles and the factors that control them. To this end, we attempt throughout our review to identify how the behavior of particles differ and how they interact.

A comprehensive review by Wood and Armitage [15] provides information on sedimentation and human activity in riverine systems, including the nature and origin of fine sediments, processes of sediment transport and deposition, and biological problems associated with increased sediment loads. Anthropogenic effects on fine sediment dynamics and related management issues are also reviewed in detail by Owens et al. [16]. Gomi et al. [17] also give a review of suspended sediment dynamics and the biological effects of forest harvesting in small streams of the Pacific Northwest. Reviews of general geomorphic processes in small, steepleland streams in relation to forest harvesting can be found elsewhere [18–20]. We complement these previous reviews by considering new material and focusing on small, steep, forested streams in several regions. We present recent contributions to this topic, including advancements in the fields of source identification, particle storage and residence time, streambed infiltration, and biological response. We also consider the importance of scale and variability to the study of fine particle dynamics. Thus, this chapter reviews fine particle dynamics in small, forested streams in relation to human activities, focusing on four general areas: 1) fine particle supply; 2) fine particle transport and deposition; 3) biological significance of fine particles; and 4) spatial and temporal variability. General characteristics of fine inorganic and organic particles are summarized and compared in Table 1.

Before we begin, it is important to consider how small streams are defined, an issue of surprising complexity and incongruity in the literature. Small streams are often defined as 1st- or 2nd-order streams in the Horton–Strahler channel-ordering system [21]. However, classification by order usually requires the analysis of topographic maps and many small channels may be excluded if map resolution is low or canopy cover is high [22, 23]. Many alternative classification schemes have been developed, based on the dominant geomorphic or hydrologic processes [24–26] and a range of quantitative and qualitative criteria. For example, Church [27] defines a fundamental distinction between ‘small channels’, scaled by the size of individual grains, and ‘large channels’, scaled by the size of grain aggregates or structures. Montgomery and Foufoula-Georgiou [28] identify a transition from debris-flow dominated, colluvial and low gradient alluvial channels at a drainage basin area of ~1 km², suggesting an alternative small/large distinction. Problems
<table>
<thead>
<tr>
<th>Sources</th>
<th>Inorganic</th>
<th>Organic</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Internal</strong></td>
<td>Bank/bar erosion</td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td></td>
<td>Bed interstices</td>
<td>Zoo- and phytoplankton</td>
</tr>
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<td></td>
<td>Surficial deposits</td>
<td>Feces</td>
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<td></td>
<td>Backwater areas</td>
<td>Biotic decay</td>
</tr>
<tr>
<td></td>
<td>Pools</td>
<td>DOM flocculation</td>
</tr>
<tr>
<td></td>
<td>Log jams/large woody debris</td>
<td>Feeding byproducts</td>
</tr>
<tr>
<td></td>
<td>Vegetation surfaces</td>
<td></td>
</tr>
<tr>
<td><strong>External</strong></td>
<td>Runoff erosion (gullies, hillslopes, soils)</td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td></td>
<td>Mass movements (landslides, debris flows, earthflows, debris avalanches)</td>
<td>Allochonous inputs (leaf litter, woody debris) and mechanical breakdown</td>
</tr>
<tr>
<td></td>
<td>Atmospheric/Aeolian deposition</td>
<td></td>
</tr>
<tr>
<td><strong>Transport/deposition/storage</strong></td>
<td>Suspended particle concentration</td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td><strong>Controlling factors</strong></td>
<td>Channel morphology (stream size, storage zone size)</td>
<td>Water temperature</td>
</tr>
<tr>
<td></td>
<td>Bed composition (roughness, physical barriers)</td>
<td>Invertebrates</td>
</tr>
<tr>
<td></td>
<td>Channel or water surface gradient, flow discharge, stream power</td>
<td>Particle size and composition (density, geometry, surface charge)</td>
</tr>
<tr>
<td></td>
<td>Particle size and composition (density, degree of flocculation)</td>
<td>Season and source</td>
</tr>
<tr>
<td></td>
<td>Large woody debris</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Frequency and timing of flow events</td>
<td></td>
</tr>
</tbody>
</table>

(Continued)
Table 1: Continued

<table>
<thead>
<tr>
<th>Characteristics</th>
<th>Inorganic</th>
<th>Organic</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Hysteresis</td>
<td>Same as inorganic particles</td>
</tr>
<tr>
<td></td>
<td>Highly variable relationship with flow</td>
<td></td>
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<tr>
<td></td>
<td>Dominated by mass movements</td>
<td></td>
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<tr>
<td></td>
<td>Little low flow transport</td>
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</tr>
<tr>
<td></td>
<td>Long residence times</td>
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<tr>
<td>External influences</td>
<td>Forest harvesting</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Changes to availability</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Expose soil</td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td></td>
<td>Reduce slope stability</td>
<td>Remove canopy cover</td>
</tr>
<tr>
<td></td>
<td>Damage streambanks</td>
<td>Reduce wood delivery</td>
</tr>
<tr>
<td></td>
<td>Accelerate mass movements</td>
<td>Change species composition (leaf-litter type, decomposition rate, timing of input)</td>
</tr>
<tr>
<td></td>
<td>Reduce transpiration and interception; raise water tables and soil moisture; increase hydrologic connectivity, streamflow and delivery</td>
<td>Increase productivity (increased sunlight, temperature, nutritional quality)</td>
</tr>
<tr>
<td>Changes to hydrology</td>
<td>Reduce magnitude of high-flow events</td>
<td>Same as inorganic particles</td>
</tr>
<tr>
<td></td>
<td>Trap particles; reduce sediment discharge</td>
<td></td>
</tr>
<tr>
<td>Flow regulation</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Alter temperature and nutrient regime in turn changing instream community composition and OM processing</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Increase seston concentrations</td>
</tr>
<tr>
<td>Spawning salmon</td>
<td>Excavate buried particles</td>
<td>Same as inorganic particles plus:</td>
</tr>
<tr>
<td></td>
<td>Accelerate downstream transport</td>
<td>Carcass decay supplies POM/DOM</td>
</tr>
</tbody>
</table>
with this definition arise, however, when considering the strong influence of local climate, geology, and history on stream size, shape, and the processes that dominate. For a complete discussion of the problems associated with defining small streams, see Benda et al. [18]. For simplicity, small streams in this review are considered those with a bankfull width less than 2–3 m or catchment area <1 ha. For the most part, we restrict our review to studies of small, forested streams in mountainous regions, although limited data availability may require us to occasionally draw from studies in larger systems or geographic locations.

2 Sources, supply mechanisms, and source identification

2.1 Fine particle sources

Fine sediment and FPOM sources may be divided into two main categories: internal, or in-channel sources, and external, or nonchannel sources (Table 1). Supply mechanism, particle composition, and sediment load will vary according to source type. Internal sources and supply mechanisms include bank or bar erosion; piping from sub-surface flow; high flow mobilization from bed interstices, surficial deposits, backwater areas, log jams, or pools; and release of particles upon senescence of aquatic vegetation. Organic particles also include small biota such as zoo- and phytoplankton, biotic waste products, or biotic decay. External sources are derived from the catchment, including runoff erosion from gullies, hillslopes, and exposed soils, mass movements such as landslides and debris flows, and atmospheric deposition. Allochthonous inputs of leaves and woody debris provide an external supply of organic matter that is further broken into fine particles by in-channel biotic activity.

Fine sediment dynamics in small, steep, forested streams can differ from large rivers in several ways. Because of their close proximity to the terrestrial environment, sediment sources reflect a mix of hillslope and channel processes, including episodic release due to mass movements, bank erosion during and between floods, and release of internal sediment stored in pools formed by large bed materials or woody debris. In mountainous regions, gullies and hillslopes are the main source of sediment, supplied to the stream via mass movements or rill erosion of exposed surfaces [18, 29, 30]. Little sediment is contributed from surface erosion by overland flow [31, 32] because of the protective effect of dense vegetation cover and well-developed organic soil horizons [33]. Sediment supply occurs by two main mechanisms: debris flows and fluvial transport, but unlike higher-order streams mass movements dominate delivery to the channel.

Entirely covered by canopy, forested streams receive litter and wood inputs comparable to that of the forest floor. Because of a greater edge-to-area ratio, bankside inputs are also high, such that small streams have higher organic matter inputs per unit area compared to larger streams in the same forest type. In-channel productivity, however, is low because little sunlight penetrates the canopy and groundwater inputs reduce stream temperatures [34]. Most organic matter in the
stream is in the form of low-nutrition wood [35], though nutritional quality increases when organic matter is broken down and colonized by microbes. Fine-particulate organic matter constitutes a large portion of the particulate organic matter pool and is comprised of leaf fragments, invertebrate faeces [36], small wood fragments [37], or flocs of DOM [38]. There is a large seasonal variation in the quality of FPOM due to the type and timing of inputs [11, 39] as well as forest age and type [11]. In regions dominated by deciduous trees, organic matter input is higher during autumn leaf-drop, such that the particle composition of the sediment load may vary over the course of the year. In contrast, coniferous inputs are evenly dispersed throughout the year. Differences in vegetation may influence organic matter dynamics [40]; for example, coniferous needles decompose more slowly than deciduous leaves [41, 42], due in part to less microbe colonization, protective chemicals, and low stream water nutrients and temperature [43]. Slow decomposition may affect the rate at which FPOM is produced, thus limiting export to downstream systems. In some areas, such as the west coast of North America, particulate and dissolved organic matter are also supplied by the decomposing carcasses of spawning salmon [44]. Excavation by redd-building salmon can also release fine particles from the streambed and accelerate transport for short distances downstream [45–47].

2.2 Source identification

Identifying the nature and relative contribution of suspended sediment sources is key to constructing watershed sediment budgets [48, 49], estimating sediment yields [50, 51] and designing effective management strategies for reducing sediment pollution [52, 53]. A wide range of source-identification techniques have been developed, but due to the complexity of factors governing sediment mobilization and supply, results are often conflicting and problematic. Because suspended sediment sources are highly spatially and temporally variable, sampling schemes, limited by logistics and costs, are often insufficient to provide representative and reliable data [54]. A comprehensive review of common approaches to source identification and the problems associated with each was provided most recently by Collins and Walling [55].

The source tracing, or fingerprinting, approach has been proposed as a less problematic, more direct technique for sediment sourcing that uses a variety of diagnostic properties to characterize and link suspended sediment samples to potential source areas [56–58]. A range of physical and chemical properties may be selected, depending on watershed and potential source characteristics, such as particle mineralogy and size [59, 60], sediment chemistry [61] mineral magnetism [56], or environmental radionuclides [58, 62, 63]. Advancements in fingerprinting techniques, including the use of multiple diagnostic properties [13, 58, 64], quantitative mixing models, and discriminant statistical tests [64–66], have enabled researchers to determine the relative contribution of source areas and supply mechanisms in many lowland and some upland rivers. A detailed review of the development and application of source fingerprinting techniques is given
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by Walling [67]; here we provide background on the use of environmental radionuclides as source tracers to facilitate discussion of their use as tracers of in-channel sediment transport in the next section.

Several studies have used both lithogenic and fallout radionuclides to identify sediment sources and temporal changes in supply and to construct sediment budgets [62, 68, 69]. Since the 1980s, a large number of in-situ produced long-lived cosmogenic radionuclides, such as beryllium-10 ($^{10}$Be) and aluminum-26 ($^{26}$Al) have been used to measure surface-exposure ages, erosion rates, and regolith production [70–72]. Long-lived lithogenic radionuclides, produced in-situ from the uranium-238 ($^{238}$U) and thorium-232 ($^{232}$Th) decay series, have also been used in fluvial systems as source tracers, linking hillslope and channel processes [73–75]. Shorter-lived fallout radionuclides, including caesium-137 ($^{137}$Cs), have been used to quantify sediment erosion, mobilization, transport and storage over shorter time scales and even individual events [63, 76–78].

Radionuclide activity of sediment is an advantageous diagnostic property because it is independent of geology and can be used to differentiate between surface and subsurface soil, as well as cultivated and uncultivated soil. Longer-lived $^{137}$Cs and $^{210}$Pb have been used to calculate decadal sedimentation rates [79], biological mixing [80], and soil erosion [81], while the short-lived $^7$Be has been used to quantify sediment transport [63, 77, 82], resuspension [83, 84], and deposition [85, 86] at event- to month-long time scales.

Additional background and details of the source tracing technique for fallout radionuclides can be found in several papers [81, 87] and in the chapter of this book entitled “Effects of land-use changes and groundwater pumping on salt water intrusion in coastal watersheds” [258]. Essentially, the approach begins by assuming that radionuclide fallout is uniform across the landscape; upon deposition, radionuclides strongly sorb to fine particles [88–90], thus movement of these radionuclides through the watershed reflects the mobilization of soil and sediment. By comparing radionuclide activity of a sample with an undisturbed reference, rates of erosion and deposition can be estimated [68]. Reference activity can be determined from direct measurement of radionuclide delivery, analysis of precipitation samples, or collection of soil or snowpack cores [14, 63, 68, 77, 91]. Atmospheric $^{210}$Pb and $^7$Be are derived naturally, but $^{137}$Cs fallout results from nuclear weapons testing in the 1950s and 1960s; all enter the ecosystem primarily through wet deposition [92–94]. Some of the variability in precipitation-delivered radionuclides can be corrected through the use of the $^7$Be/$^{210}$Pb ratio [77, 95, 96]. Because fallout radionuclides are atmospherically derived, activity typically declines exponentially with soil depth [13, 58, 68], allowing differentiation between surface and subsoil-derived sediment. In addition, because plowing and tilling of soil mixes high-activity surface soil into lower layers, sediment derived from the surface of cultivated soils will have lower activity levels than uncultivated soils. Each radionuclide distributes differently in the soil, thus sediment source areas can be distinguished by the relative amounts of different radionuclides corresponding to land use and depth [14, 97, 98]. Furthermore, upon entering the river, sediment is
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no longer exposed to the atmosphere and the radionuclide activity begins to decay. Thus the activity of fluvial sediment reflects mixing between landscape-, bank- and streambed-derived sediment, storage times within the channel, and transport rates through the river system. Application of fallout radionuclides to channel processes such as transport and deposition will be discussed further in the next section.

2.3 Impact of human activities on sources

Due to the spatiotemporal variability of sediment sources, human activities that increase fine sediment input to streams are often regulated as non-point sources of pollution [99]. Thus, techniques that improve our ability to accurately identify the location of dominant sediment sources, and thus the activities associated with them, will greatly aid management efforts. Small, forested watersheds are particularly impacted by forest harvesting practices, such as logging, road building and slash burning, which can both indirectly and directly affect the sources and supply of sediment to streams (Table 1). The main impact of forest harvesting activities is an increase in sediment availability; all activities disturb and expose soil, alter slope stability, and damage streambanks, increasing sediment mobility from surface sources and destabilizing stored sediment. In particular, mass movements are accelerated post-harvest due to reduced hillslope stability when roots are removed from streamside areas [100, 101]. Both mass movements and surface runoff from logging roads can increase sediment load to stream channels [102, 103], due in part to construction on unstable terrain and poor road drainage [104]. Although some studies have shown that roads can also act as depositional and storage sites for sediment [105], roads generally increase sediment production and input to fluvial systems, depending on hillslope position, timing of logging activity and large storms, and road-management practices [104–107].

Tree removal also reduces transpiration and interception, increasing soil moisture and water table levels [108, 109], which can increase connectivity between perennial and ephemeral streams and subsequent sediment delivery. Despite extensive study, uncertainty still remains over the relative influence of hydrologic changes versus increased sediment supply following harvesting. In particular, sediment yield to headwater streams appears to increase due to changes in flow rather than sediment supply [110, 111], but this is largely unexplored.

Tree removal from streamside areas also alters the type, amount, and timing of organic matter delivered to streams by altering tree-species composition, removing canopy coverage, and reducing wood delivery. The transition from old growth conifers to young, deciduous regrowth changes the type of organic matter inputs – from needles to leaves – and the timing of inputs – from year-round to seasonal [10]. Riparian buffers are commonly proposed as a strategy for intercepting sediment input via overland flow and minimizing physical disturbance adjacent to the stream. In most cases, riparian buffers reduce increases in sediment yield following harvesting (see Gomi et al. [17] for a review) but are ineffective at intercepting sediment generated outside the riparian zone or by mass movements and road erosion [17, 112, 113].
Table 2: Characteristics of three theoretical models representing fine particle distribution and movement that differ in application, approach and intended focus.

<table>
<thead>
<tr>
<th>General form</th>
<th>Rouse equation</th>
<th>Advection-dispersion model</th>
<th>Local Exchange Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vertical solution (1-D)</td>
<td>$C_z = C_z \left[ \frac{z}{z_0} \left( \frac{h-z_0}{h-z} \right)^{S} \right]$</td>
<td>Continuity equation (3-D)</td>
<td>$\frac{\partial f}{\partial t} + \frac{\partial}{\partial z} \left[ (\mu - \frac{1}{2} v^2) f \right] = 0$</td>
</tr>
<tr>
<td>Continuity equation (3-D)</td>
<td>$\frac{\partial c}{\partial t} + \nabla \cdot (cu - K \nabla c) = 0$</td>
<td>Stochastic-diffusion equation</td>
<td>$\frac{\partial f}{\partial t} + \frac{\partial}{\partial z} \left[ (\mu - \frac{1}{2} v^2) f \right] = 0$</td>
</tr>
<tr>
<td>Longitudinal solution (1-D)</td>
<td>$\frac{\partial c}{\partial t} = -\frac{Q}{A} \frac{\partial c}{\partial x} + \frac{1}{A} \frac{\partial}{\partial x} \left[ AK(x)C \right]$</td>
<td>Normalized stationary vertical profile (1-D)</td>
<td>$\frac{d}{dz} \left[ u_c^* - K \frac{dc^*}{dz} \right] = 0$</td>
</tr>
<tr>
<td>$+ \frac{Q}{A} (C_L - C) + a(C_S - C)$</td>
<td>and</td>
<td></td>
<td></td>
</tr>
<tr>
<td>$\frac{\partial c_s}{\partial t} = -\frac{A}{A_s} (C_S - C)$</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Variables:

- Rouse number: $S = \frac{V_{rel}}{\beta k u_c}$
- Concentration gradient vector ($\nabla c$)
- Advection velocity vector ($u$)
- Matrix of longitudinal, transverse, and vertical dispersion coefficients ($K$)
- Infinitesimal variance and mean in vertical direction

$\nu(z) = 2K(z)$
$\mu(z) = u_c(z) + K'(z)$

(Continued)
Table 2: Continued

<table>
<thead>
<tr>
<th>Rouse equation</th>
<th>Advection-dispersion model</th>
<th>Local Exchange Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Particle fall velocity ( V_{fall} )</td>
<td>Discharge ( Q )</td>
<td>Vertical dispersion rate</td>
</tr>
<tr>
<td>Particle diffusivity ( \beta )</td>
<td>Cross-sectional area of the stream ( A )</td>
<td>( K(z) = \psi \left[ \frac{M}{2} + \sqrt{\frac{M}{2}} \right] + l(z)^2 u_\ast^2 (1 - z / H) )</td>
</tr>
<tr>
<td>von Karman’s constant ( \kappa )</td>
<td>Diffusion coefficient in the downstream direction ( K(x) )</td>
<td>Constant &gt; 0 ( \psi )</td>
</tr>
<tr>
<td>Bed shear velocity ( u_\ast = \sqrt{\tau_0 / \rho} )</td>
<td>Groundwater or tributary inflow ( Q_L )</td>
<td>Kinematic molecular viscosity ( M )</td>
</tr>
<tr>
<td>Bed shear stress ( \tau_0 )</td>
<td>Solute concentration of inflow ( C_L )</td>
<td>Mixing length (measure of kinematic eddy viscosity) ( l(z) )</td>
</tr>
<tr>
<td>Near-bed concentration ( C_o )</td>
<td>Area of the transient storage zones ( A_s )</td>
<td>First derivative of ( K(z) ) ( K'(z) )</td>
</tr>
<tr>
<td>Reference elevation ( z_0 )</td>
<td>Concentration of solute in the transient storage zone ( C_s )</td>
<td>Vertical component of the advective velocity ( u_z(z) ) (= –s(z), vertical component of ( V_{fall} ) )</td>
</tr>
<tr>
<td>Water depth ( h )</td>
<td>Coefficient of exchange with the transient storage zones ( \alpha )</td>
<td>Normalized steady-state concentration ( c^* )</td>
</tr>
<tr>
<td>Elevation ( z )</td>
<td></td>
<td>( c^* (z) = \frac{c(z)}{\int_0^z c(\zeta) d\zeta} )</td>
</tr>
<tr>
<td>Concentration at ( z \left( C_z \right) )</td>
<td>[ \begin{align*} K(z) = \psi &amp; \left[ \frac{M}{2} + \sqrt{\frac{M}{2}} \right] + l(z)^2 u_\ast^2 (1 - z / H) \end{align*} ]</td>
<td>Particles of uniform size, shape, density</td>
</tr>
</tbody>
</table>

Assumptions
- 2D steady, uniform flow
- Particles of uniform size, shape, density
- Flat, hydrodynamically smooth bed
- Particles of uniform size, shape, density
- No particle–particle interactions
- Flat, hydrodynamically smooth bed
Straight channel
Mean particle size used in computation of $v_{\text{fall}}$
Depth-independent variables ($\beta$ and $\kappa$)
No particle–water or particle–particle interactions

Downward velocity of particles equals particle fall velocity
Particle diffusivity equals turbulent momentum diffusivity of water

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**Application**
- Inorganic particles
- Theoretical studies

**Dimensions**
- 1D (vertical)
- Cross-sectional scale

**Advantages**
- Simple; few parameters

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No particle–water or particle–particle interactions
Longitudinal particle velocity equals local water velocity
Downward velocity of particles equals particle fall velocity
Particle diffusivity equals turbulent momentum diffusivity of water

**Principle of local exchange:** upward and downward movements of water always balance (no net transport)

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**Application**
- FPOM
- Field studies

**Dimensions**
- 3D (usually only longitudinal)
- Reach scale

**Advantages**
- Incorporate stream morphology, groundwater and tributary inputs, transient storage, and immobilization of nonconservative solutes
- Estimate extent of transient storage and particle exchange

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Downward velocity of particles ($uz(z)$) equals particle fall velocity

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FPOM; living, motile organisms
Theoretical, field, or flume studies
3D (initially only vertical)
Particle scale
Consider individual particle movement as stochastic-diffusion process
Incorporates effect of flow on particles with depth-dependent vertical dispersion coefficient ($K(z)$)
Includes motile particles
Provides equations for concentration profile
Provides equations for probability distributions of particle-hitting time and hitting distance and dependence on initial elevation and fall velocity

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(Continued)


<table>
<thead>
<tr>
<th>Limitations</th>
<th>Rouse equation</th>
<th>Advection-dispersion model</th>
<th>Local Exchange Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flows will distribute particles nonuniformly due to differences in fall velocity</td>
<td>Particle–water interactions influence turbulent flows, streamwise velocities, and particle diffusivity</td>
<td>Particle–water interactions influence turbulent flows and streamwise velocities</td>
<td>Natural beds are not perfectly smooth</td>
</tr>
<tr>
<td>Natural beds are not perfectly smooth C&lt;sub&gt;d&lt;/sub&gt; difficult to measure</td>
<td>Biological influences only represented indirectly Streamwise velocities vary with depth Downward particle velocities do not always equal fall velocities Particle diffusivity varies with depth, density of suspended particles and bed conditions</td>
<td>No terms for morphology, flow inputs, transient storage, or non-conservative solutes</td>
<td>Purely physical model; no biological influences (except motile particles) Downward particle velocities do not always equal fall velocities</td>
</tr>
<tr>
<td>Particle–water interactions influence turbulent flows, streamwise velocities, and particle diffusivity</td>
<td>Particle diffusivity varies with depth, density of suspended particles and bed conditions</td>
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</tr>
</tbody>
</table>

References:

Bagnold [236], Einstein and Chien [122], Halbrom [237], Hunt [238], Rouse [239], Tanaka and Fugimoto [240]

Bencala and Walters [242], Valett et al. [243], Webster and Ehrman [244], Cushing et al. [152], Minshall et al. [113], Newbold et al. [159], Paul and Hall [148]

McNair et al. [175], McNair [158], McNair and Newbold [159]
3 Particle transport, deposition, and streambed infiltration

Upon entering the fluvial system, fine particles may be transported downstream or deposited on the surface of the streambed. Once deposited, these particles may be retained, accumulating or infiltrating into the bed, consumed, or entrained back into the water column. A number of physical and biological factors determine the fate of fine particles (Table 1), including suspended particle concentrations [114], bed composition [115], channel morphology [116, 117], benthic ecology [118], particle size and composition [119, 120], and flow discharge [121, 122]. For decades, both physical and ecological researchers have quantified and modeled particle transport and deposition, greatly enhancing our understanding of particle movement and storage. A high degree of variability and uncertainty among results, however, arises from the inherent complexity of factors that govern particle dynamics.

Most simply, the mode and rate of particle transport is a function of particle size, density, and stream discharge. Generally, fine particles such as silts, clays, or FPOM are carried in suspension; therefore, under normal flow conditions they are unlikely to frequently interact with the bed or have long residence times within the stream. In contrast, larger particles roll and saltate along the bed as bed load [123] and have longer residence times. The boundary between these two transport modes is transitional; depending on flow magnitude, medium and coarse sand (0.25–2 mm) may move either short distances in suspension or roll along the bed as bed load. In practice, however, these physically based distinctions are blurred. Because of instrumental and practical limitations, measurements of sediment transport are divided into bed or suspended load; transitional material is included in one or the other depending on flow and channel conditions. Under typical flow conditions, more than 90% of suspended load is composed of very fine particles such as silt and clay, whereas particles 0.25–2 mm (bed material load) are retained in the channel bed. Numerous physical models and measurement techniques have been developed to quantify suspended and bed-load transport [124–127]. Further information on the long history of this field can be found in several papers, including Hassan et al. [19].

Focusing primarily on small, steep streams, we first review recent studies of fine particle transport and deposition that attempt to address the complexities inherent in these processes, including ecological studies concerned with the movement and storage of FPOM. Secondly, we discuss recent insights into the mechanisms of streambed infiltration and retention, a topic of increasing interest due to its potentially serious biological consequences. Thirdly, we present methods of tracing particle transport used in the physical and ecological sciences, highlighting recent advancements into the use of fallout radionuclides to track particle movement. Fourthly, we introduce three theoretical models of particle movement and vertical distribution that differ in their application, approach and intended focus (Table 2). Lastly, we close this section by evaluating how human activities in forested watersheds change in-channel particle dynamics.
3.1 Fine particle transport and vertical movement in the water column

Most fine particle transport from small streams occurs during snowmelt periods and single floods, usually of short duration and high magnitude, displaying greater inter-event and intra-event variability than lowland streams [122] and a highly variable relationship between suspended load and water discharge. Transport is highly related to discharge and often exhibits hydrograph hysteresis, with concentrations at a given flow on the rising limb much greater than the corresponding flow on the falling limb [30, 128, 129]. Patterns of hysteresis in the relation between suspended load and water discharge are related to types and locations of active sources [30, 122]. High-flow events also change the transport, storage, and characteristics of organic particles. Lateral flooding can deposit significant amounts of organic matter onto floodplains and banks, reducing overall export [130]. Extensive mass transfer and exchange of organic matter can occur during flood events, increasing or decreasing surficial organic particles [131–135], depending on organic matter storage and load, as well as the retention capacity of a given section of streambed. Organic content, bioavailability, and metal affinity also differ between particles generated during a flood event and those accumulated during low-flow periods [136], with possible nutritional effects on the growth and metabolism of benthic organisms.

Particle deposition and storage in small streams occurs primarily in pools associated with physical barriers in the stream, so that total storage capacity is less than downstream alluvial reaches with floodplains, bars, and side channels. Large structural elements such as boulders and large woody debris control the amount of particle storage and channel stability [137, 138]. Wood and inorganic substrates form a high degree of roughness relative to stream depth that forms steps, pools and depositional areas, influencing the movement of inorganic and organic particles [130, 139, 140]. Episodic mass movements such as landslides and debris flows dominate transport in these systems [141], with little to no transport during low flows. As a result, these streams may act as inorganic and organic particle reservoirs for long periods of time, but the timing and frequency of storage release is highly unpredictable. Thus, periods of particle storage may be much longer and transport more temporally variable in small streams than in larger channels.

Ecologists have long been interested in the factors controlling FPOM transport and storage because of its implications for ecosystem productivity and diversity. Most studies of FPOM dynamics in streams focus primarily on longitudinal (up- to downstream) linkages and the quantification of particle transport distance ($S$), depositional velocity ($V_{dep}$), and residence time. These studies have elucidated many of the factors that govern FPOM dynamics, including bed and channel roughness [7, 115]; channel gradient, flow discharge, and stream power [115, 121, 142–144]; water temperature [145]; debris dams [146]; invertebrates [118, 147, 148]; stream and storage zone size [115]; the frequency and timing of flow events [7, 128, 148]; and particle size, geometry, surface charge and density [149, 150], in turn a function of organic matter season and source (i.e. allochthonous vs. autochthonous) [151, 152].
Downward movement through the water column and the rate of streambed deposition can be quantified by particle depositional velocity ($V_{\text{dep}}$), the rate at which a released particle settles on the bed. Depositional velocity is primarily controlled by the particle’s still-water settling velocity ($V_{\text{fall}}$), which is in turn a function of particle size, shape, and density [153]. In theory, the measured $V_{\text{dep}}$ should be approximately equal to $V_{\text{fall}}$, but most field studies show no consistent relationship between these two parameters [115, 154–156]. The ratio of $V_{\text{dep}}$ to $V_{\text{fall}}$ in these studies ranges from 0.04 to 1690, depending on particle properties and stream characteristics, but generally increases with decreasing particle size and density. Fluorescently labelled bacteria has the highest ratio (1690) [155] while natural FPOM and corn pollen are much lower (0.04–0.56).

In the cases where $V_{\text{dep}}$ is $<< V_{\text{fall}}$ [115, 144, 150], it is proposed that turbulent mixing and resuspension factors overwhelm gravitational settling in controlling particle movement. This is represented in theoretical models that predict $V_{\text{dep}} < V_{\text{fall}}$ when local shear stresses exceed a critical threshold for resuspension [157]. The ratio of gravitational velocity to turbulent mixing velocity (due to bed shear) is expressed by the Rouse number ($\hat{s}$). There is little effect of gravity at values of $\hat{s} < 0.1$ but gravitational factors increasingly dominate as $\hat{s}$ approaches 1 [158, 159]. Georgian et al. [160] calculate $\hat{s} < 0.01$ and low $V_{\text{dep}} < V_{\text{fall}}$ for both FPOM and pollen in the field and $\hat{s}$ approaching 0.1 for FPOM in a flume, demonstrating the importance of shear forces over gravity in the field and increased importance of gravitational settling in a flume. A large $V_{\text{dep}}$ may occur in situations where energy at the streambed is dissipated by plunge pools or stagnant zones, therefore decreasing bed shear stress and turbulent resuspension. Minshall et al. [115] report $V_{\text{dep}} \sim V_{\text{fall}}$ in small streams with high bed complexity, while Georgian et al. [160] report that $V_{\text{dep}}$ in a smooth-bedded flume is less than $V_{\text{dep}}$ of the same particles in the field, both suggesting that bed complexity and its effect on turbulence may strongly influence the deposition of particles. Particles with very low settling velocities and larger depositional velocities (i.e. bacteria) may also be deposited via advective transport into interstitial spaces, hyporheic entrainment, or adhesion to the substrate.

Traditional models explain the discrepancy between $V_{\text{dep}}$ and $V_{\text{fall}}$ by the hydrodynamic/gravitational mechanisms described above, but so far, measurements of bed roughness and shear stress have not revealed a consistent or significant relationship with $V_{\text{dep}}$ [115]. Furthermore, conflicting and limited results have not fully elucidated the mechanistic role of transient storage zones in FPOM deposition. For example, Minshall et al. [115] report that $V_{\text{dep}}$ was positively correlated with the relative size of transient storage zones and a coefficient of transient storage exchange, but not related to the advective exchange of water into these zones. Paul and Hall [150] report no relationship between $V_{\text{dep}}$ and any measure of transient storage or exchange. Newbold et al. [161] measure brief retention of FPOM at a rate similar to that of water retention in transient storage zones, indicating simple advective transport to and from these zones without deposition. Based on earlier findings that hyporheic exchange increases particle removal from the water column [162], Newbold et al. [161] propose that most transient storage may not
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occur in hyporheic zones, but rather in deep lateral areas where turbulence is high enough to keep particles suspended. Differences between studies may be due to stream-to-stream variation, the size and composition of particles, the relative amount of in-channel versus hyporheic zone storage, or biological properties of the streambed. Newbold et al. [161] also calculate that average residence times of deposited particles are longer than turbulent fluctuations, suggesting the influence of biological retention processes. Evidence for the role of biological mechanisms in particle retention is limited, but suggestive. For example, biofilm adhesion is proposed as one explanation for the discrepancy between $V_{\text{dep}}$ and $V_{\text{fall}}$ [163, 164], while invertebrate manipulation [148] or removal by filter feeders may account for a measurable proportion of total deposition [131, 165, 166]. Particle composition may also explain much of the observed discrepancy, an explanation that has received increasing attention in the past decade. As noted above, most suspended particles are composites of organic and inorganic components. Composite particles may enter the stream from the watershed as aggregates, retaining their structure during transport, or may form in-channel via physical, biological, or chemical flocculation processes [176–178].

Cohesive properties of organic matter tend to enhance flocculation [119, 120]; however, some evidence exists for electrochemical flocculation in glacial meltwaters that lack organic matter [178]. Particle size and hydrodynamic properties of composite particles differ considerably from their mineral components. Thus, predictions of deposition rates assuming single-grain settling are likely to be inaccurate when composite particles predominate [176–179].

3.2 Fine particle deposition, retention and infiltration in the streambed

Rates and mechanisms of particle deposition, retention, and streambed infiltration determine the type and degree of impact fine particles will have on benthic ecology, including biotic metabolism, nutrient cycling [180], and contaminant delivery [8, 181–185]. Collectively termed streambed clogging, particle deposition and infiltration into the hyporheic zone reduces interstitial habitat and the exchange of water and solutes with subsequent effects on benthic community structure. Biological consequences of clogging are discussed further in the next section and are given a full review by Gayraud et al. [186].

Retention of inorganic and organic particles can occur by several physical mechanisms, including filtration, sedimentation, burial, and vertical hydraulic entrainment. On an armored bed, small sand grains may infiltrate the pore spaces of the bed surface, moving only when the entire bed is entrained [116]. Filtration into hyporheic zone sediments varies spatially and temporally, depending on geomorphic and hydraulic conditions of the streambed, in turn forming patches of differing exchange capacities and transient storage characteristics [187]. Sedimentation of particles occurs in regions of low flow, such as pools, eddies or backwaters, created by structural elements within the channel [154, 188–190]. High sedimentation and low resuspension rates produce zones of accumulation. Burial by shifting sediments occurs in lowland, sand-bedded
rivers when migrating bedforms bury organic particles deposited in the lee of the structure [191–193]. Vertical hydraulic entrainment occurs in well-sorted, open-frame sediments, where local differences in water pressure produce vertical hydraulic gradients of up- and downwelling water, mixing materials between the surface and interstitial water. Several studies have documented vertical exchange in riffle-pool sequences [194–197], where even single boulders can increase vertical interactions [198] and FPOM concentrations in the hyporheic zones of riffles [199, 200]. Bed-roughness elements, whether inorganic or organic, can produce vertical fluxes of water and materials between the stream and bed sediments [192, 193, 201], with ensuing affects on nutrient and carbon dynamics. Downward fluxes of oxygenated water can maintain aerobic conditions deep within the sediment. Combined with the flux of organic particles, this phenomenon contributes to the entrainment, storage, and microbial transformation of these particles within bed sediments.

3.3 Measuring fine particle transport and infiltration

Because of their small size and stochastic behavior, tracking the movement and storage of individual fine particles has until recently, remained elusive. Previous studies primarily focused on the mode of fine sediment transport; whether fine particles are transported in a series of steps and jumps [202–205] or travel the length of a river in a single hydrograph [206, 207]. However, several recent studies have used radionuclides to calculate downstream transport distances and channel-bed mixing of suspended sediment [66, 77, 82, 208] and transitional bed material [209]. Using $^7$Be, Bonniwell et al. [77] found that suspended fine particles have short residence times and travel long distances during high flow periods in high gradient streams. Using the activity of $^7$Be, $^{137}$Cs, and excess $^{210}$Pb in source soils and suspended sediment, Whiting et al. [63] found that the radionuclide signature of suspended sediment reflects the relative contributions of upland soil and bank erosion and that transport distances increased with basin size. Both studies calculated transport distances from the exponential decrease in activity from source areas to suspended sediment samples downstream. Alternatively, the radionuclide signature of particles can be used to track movement through the channel or watershed. For example, Salant et al. [209] tracked the movement of a pulse of sediment released from storage behind a dam, where it had become depleted in $^7$Be, and thus determined sediment transport velocities. Results from these studies demonstrate that radionuclides can be effectively used as fine particle tracers for determining transport and deposition rates.

Meanwhile, in the field of ecology, more than two decades of research has led to the development of a range of field techniques for tracking the source and movement of FPOM. In most cases, a known concentration of particles, either $^{14}$C-labeled natural FPOM [115, 156, 160, 210, 211] or a FPOM analog or surrogate, is released into the stream at a fixed point and the concentrations of water samples are measured at subsequent downstream stations. These concentrations are used in
an exponential decay model to determine the loss of particles from the water column with distance [144, 150, 154, 211–213]:

\[ N_x = N_0 e^{-kx}, \]  

(1)

where \( N_x \) and \( N_0 \) are the quantities in suspension at a distance \( x \) and at the point of introduction, respectively, and \( k \) is the longitudinal loss rate of particles. Average transport distance \( (S) \) is the inverse of \( k \). The paired release of a conservative tracer allows for the calculation of flow discharge. In order to correct for differences in water depth and velocity that may exist between streams, transport distances are converted to a related parameter, deposition velocity \( (V_{dep}) \) [152, 148, 142], by the equation

\[ V_{dep} = hV_{water}k = \frac{hV_{water}}{S}, \]  

(2)

where \( h \) and \( V_{water} \) are the depth and velocity of water, respectively. The mean time in suspension \( (T) \) is estimated as \( S/V_{water} \).

Fine sediment infiltration can be measured using coring and pumping techniques or sediment collectors. Sediment collectors, also known as infiltration traps, only measure fine sediment accumulation, whereas coring and pumping techniques are also used to collect chemical and invertebrate samples. Although quantitatively the most robust and reliable, coring techniques are typically complex, time-consuming, labor-intensive, and expensive. In small streams dominated by coarse particles, freeze cores are most commonly used. Freeze-coring techniques are ideal for quantifying vertical variability in subsurface material because they allow for the removal of a vertical section of streamed that can be further divided by depth. Core installation generally follows a standard protocol [214], in which the evaporation of liquid carbon is used to freeze sediments around a probe inserted into the bed. Upon removal, cores are typically subdivided and analysed for particle-size distribution and organic matter content. Problems with this technique do exist, however, including bed disruption [215], bias due to an irregular sample boundary [216], and small sample sizes that result in high variability among individual cores, bedforms, and reaches. Previous studies have shown that a minimum sample size with total weight of 20 kg, including 5 cores from each sample site, provides reproducible results and a subdivision depth of 15 cm reduces the error in subdividing cores [214, 217–219]. Furthermore, biases of the freeze-core technique are negligible when only a sample of the fine matrix fraction is sought [214, 217, 219].

Less intensive and less expensive than coring, the Bou–Rouch pumping method is commonly used for sampling hyporheic water chemistry, sediment, and invertebrate fauna. In this method, water is pumped from the streambed at several depths using temporary or permanent standpipe wells [220]. Although biases may occur towards invertebrate types due to sample volume, sediment filtration, well permanence, attraction of predators, recovery time of fauna, and pumping rate [169, 221–224], the sampling of fine sediment content by this method produces similar results to coring [225]. Other concerns exist, however: well installation, shape, and size can modify substrate conditions [226], pumping cannot collect particles >1 mm [225],
and streambed location can be uncertain in cases of high sediment heterogeneity (Palmer and Strayer, [220]).

Sediment collectors or infiltration traps offer the simplest, least expensive method for assessing streambed infiltration. Several varieties of traps exist, but most are composed of a solid-base, mesh container filled with clean gravel. Traps are embedded for a period of time, usually over the course of one or more flow events, then removed and analysed for fine sediment content [216, 227–234]. In addition to assessing the amount and characteristics of sediment collected, traps have also been used to determine controls on infiltration, such as suspended and bed-load transport rates or the relative size of transported and streambed sediment [227, 229, 235–239]. Detailed descriptions of commonly used instruments can be found in several papers [216, 230, 233]; two recently developed methods are briefly described here. Lachance and Dube [240] designed a collector from two solid-based, 1–L cylindrical buckets set inside each other, drilled with multiple 1–3 cm matching holes to allow the flux of water and sediment, and sealed before removal by rotating the outer bucket. Although these traps appear adequate for assessing and comparing infiltration rates and sediment types, discrepancies between sediment accumulation into collectors and natural salmon redds indicates that these traps may overestimate actual streambed infiltration, possibly due to lower sand content and large pore volume [234]. Newly developed by Levasseur et al. [241], the infiltration cube is proposed as an efficient and reliable instrument for sampling large (~65 kg) volumes of bed sediment, with minimal bias to infiltration rates. Modified from the previously developed infiltration bag [216], the cube is formed of a rectangular metal frame with a folded plastic bag at its base. Installation and burial of the cube is designed to mimic redd construction of a spawning salmon, producing a structure with grain size and morphological characteristics typical of natural redds. Four wires at the corners of the bag protrude from the streambed to raise the bag around the frame upon removal, ensuring retention of all collected material. Although intended for assessments of salmonid spawning habitat and the effect of fine sediment on embryo survival, this technique may be applied to studies of sediment dynamics by modifying installation procedures and filling the cube with gravel of a known composition and porosity. As yet, only a single published study has used this instrument, demonstrating successful retention of even the finest particles upon removal of the sample, permitting the assessment of very small-scale variations in the amount and effect of this size fraction [242]. Further studies for a range of environments and applications are needed to fully test this technique.

3.4 Models of vertical particle distribution and exchange

Although early studies of FPOM dynamics focused primarily on longitudinal transport, ecologists are increasingly recognizing the importance of vertical particle exchange, particularly rates of particle deposition, retention, and resuspension, to stream functioning. Three models are introduced and briefly described here: the Rouse equation, the advection-dispersion model, and the Local Exchange Model; these are summarized and compared in Table 2. Limitations of these three models are also discussed.
3.4.1 The Rouse equation

In the physical sciences, decades of research has led to the development of robust theoretical equations for the vertical distribution of suspended particles in water, based primarily on mathematical representations of turbulent flow, molecular diffusion, and particle movement. Although confirmed by early studies in uniformly turbulent tanks known as “turbulence jars” [243–245], these equations rely on empirically derived constants and several assumptions that may not be valid under natural conditions. Several such equations exist [124, 167–171, 246], each attempting to improve those developed before, but all containing the same fundamental model, basic assumptions, and empirical constants. Rouse [170] developed the original form from a differential equation of suspended sediment (Table 2) that assumes two-dimensional, steady, uniform flow and particles with uniform size, shape, and density. Steady flow implies that the average concentration of particles remains constant, such that the flux of particles in all three dimensions is balanced; in other words, for a given volume, sediment fluxes in and out are equal. Thus, particle distribution in the vertical direction represents a balance between the downward settling of particles due to gravity and the upward movement of particles due to diffusion [247].

However, despite the theoretically robust nature of this model, it is difficult to apply in practice because near-bed concentration (\(C_a\)) and the Rouse number (\(\hat{s}\)) (Table 2) are poorly constrained, particularly over beds with a wide range of particle sizes. In theory, both \(C_a\) and \(\hat{s}\) are dependent on the flow level and the particles available for transport, which is in turn a function of bed composition. Previous work has used a mean particle size for the computation of particle fall velocity in the equation for \(\hat{s}\), despite the fact that given a wide range of particle sizes and densities, flows will distribute particles non-uniformly according to their fall velocities; flows will selectively transport fine particles higher in the water column, particularly as flow intensity decreases. The Rouse number at any point in the profile may differ depending on the grain size and density of suspended particles. Variation in sediment diffusivity (\(\beta\)), normally assumed constant, can also change the value of \(\hat{s}\). Several studies have measured suspended sediment profiles and demonstrated wide variation in sediment diffusivity depending on the region of the flow depth [248, 249], the density of suspended particles [114], and the conditions of the bed [250]. These discrepancies indicate that the Rouse equation may be inappropriate for application to most natural systems.

3.4.2 The advection-dispersion model

Similar to the Rouse equation, the advection-dispersion model [172–174] describes solute or particle flux in three spatial dimensions as a balance between the transport driven by particle fall velocity (\(V_{\text{fall}}\)) and flow velocity (advection) and that driven by molecular diffusion and turbulence (dispersion) (Table 2). In this framework, the concentration of an assemblage of particles is modeled as a deterministic diffusion process. Integration of the advection-dispersion model over the flow depth produces an estimate of the average concentration of suspended particles.
The advection-dispersion model can be expanded to include other stream characteristics such as morphology, groundwater and tributary inputs, and transient storage [174] (Table 2). Transient, or temporary, storage of a solute or particle can occur in slower moving areas of the stream, such as pools or hyporheic zones [172]. In the case of a nonconservative solute, abiotic and biotic exchanges take place between the water column and the stream substrate (i.e. adsorption, plant uptake). Immobilization, the removal of solutes from the water column, can be incorporated into a model of nonconservative solute dynamics with an additional term. Terms within the model that represent transient storage and immobilization describe the interaction between the free-flowing stream channel and hydrologic storage zones. A number of studies have used this model to investigate the relationship between particle dynamics and the extent of transient storage [115, 150, 154, 161].

Prediction of suspended sediment transport by the advection-dispersion model requires three major assumptions: 1) the streamwise particle velocity is equal to the local water velocity, 2) the downward velocity of particles in turbulent flow is equal to the particle fall velocity, and 3) particle diffusivity is equal to the turbulent momentum diffusivity of water [114]. However, these assumptions ignore the small-scale mechanics of suspended particles, implying that particles and water act as a single phase or mixture. As a result, predictions based on these three assumptions may be of limited validity, because particle–water and particle–particle interactions can influence the dynamics of turbulent flows and streamwise velocities. Several studies have demonstrated that because of these interactions the above assumptions are in fact questionable [114, 251–254]. For example, using image-based techniques to separately quantify particle and water movement, Muste et al. [114] measured streamwise particle velocities less than the velocity of water in the upper region of the flow and a reverse relationship in the lower region, violating the first assumption. They also found that vertical velocities of sand particles differed from water velocities throughout the flow depth. Downward vertical velocities of sand particles were lower than water velocities in the upper and near-bed regions, but greater in the lower flow region. Although this study did not compare these velocities directly to particle fall velocities, the variation in downward velocities contradicts assumption two. This study also showed that sediment diffusivity was less than that of water in the mid- and upper regions of the flow, contradicting assumption three. These findings demonstrate that the advection-dispersion model is limited because it does not consider particle–water interactions or the depth-dependent nature of model parameters.

3.4.3 The Local Exchange Model

McNair et al. [175] propose an alternative approach to modeling suspended particle dynamics that considers the behavior of an individual particle, rather than an assemblage of particles. Like the advection-dispersion model, the focus and intended application of the McNair et al. [175] approach is FPOM. The authors argue that theoretical models developed for suspended inorganic particle dynamics are inadequate for ecological applications because of their focus on inorganic
particles with high fall velocities, passive gravitational settling, and physical forces, as well as an interest in large-scale processes. In contrast, ecological studies emphasize the biological significance of particles that often vary in shape, composition, and density and may be deposited and mobilized via behavioral means. McNair et al. [175] describe the process of fine particle transport as including four key components: the attachment problem, the entrainment problem, the hitting-time problem, and the hitting-distance problem. The attachment and entrainment problems address how a particle at the bed/water interface becomes fixed to the substrate and its residence time before resuspension into the water column. The hitting-time/distance problems consider the temporal and spatial dimensions of longitudinal transport; namely, how long a particle remains in the water column and the distance it travels. Time spent in the water column may be relevant to free-swimming consumers, while travel distance is important in determining the rate of downstream dispersal.

Fine particles in turbulent water move along irregular trajectories, buffeted up and down by fluid eddies, thus vertical particle movement and elevation may be considered a stochastic process [175, 255]. McNair et al. [175] provided a discrete representation of this stochastic process by considering the motion of a neutrally buoyant, nonmotile particle as occurring in two ways: particles can be propelled by molecular collisions or may be incidentally carried by the turbulent transport of water. Because of the complex, nonlinear structure of turbulent fluid motion, a simplified approximation, known generally as a stochastic-diffusion process, can be used to model turbulent transport. All stochastic-diffusion processes are defined by forward and backward Kolmogorov equations, which include the infinitesimal mean and variance functions (Table 2). Once specified, these functions convert the abstract stochastic-diffusion process into a meaningful description of the process of interest, i.e. the dynamics of particle motion. For more detailed explanation of the background and equations of the Local Exchange Model, see McNair [156], McNair and Newbold [157], and McNair et al. [175].

Although the Local Exchange Model improves upon the advection-dispersion model by addressing the small-scale dynamics of individual particle motion and the inclusion of depth-dependent parameters, it retains assumptions that limit the model’s validity. The Local Exchange Model does represent the effect of fluid motion on particles by a vertical dispersion coefficient that varies with depth ($K(z)$) (Table 2), but it does not consider the reverse effect of particles on the flow that can occur throughout the flow depth regardless of particle density [114]. As noted by McNair et al. [175], close agreement between vertical profiles predicted by the Local Exchange Model and data from the literature does not fully validate the model because the available empirical evidence is limited to flume data and one field case that only considered particles of high fall velocities [256–258]. Vertical profiles from field measurements and a range of particle fall velocities are needed to test the model. Furthermore, the model contains a number of assumptions and approximations that may not hold under all conditions, most importantly the assumption of a flat, hydrodynamically smooth bed. Extension of the model to a complex bed topography, where the presence of retention structures or transient
storage zones may greatly alter particle dynamics and the profile of vertical mixing, would be difficult and highly uncertain. The influence of bed topography may be greatest for small particles whose vertical movement is strongly affected by turbulent flows. It should also be noted that despite being developed within an ecological framework, the Local Exchange Model is a purely physical-based model that does not consider biological influences.

3.5 Impact of human activities on particle transport

Human activities, such as dam operation and forest harvesting, can alter the timing and magnitude of hydrologic events, altering discharge regimes and sediment transport (Table 1). Dams both store water and capture sediment, so that the downstream impact differs whether considering the dam’s effect on sediment discharge or its effect on transport capacity. Reduced sediment loads may enhance channel armorng, while reduced flows may lead to sediment deposition if the tributaries and banks contribute more sediment than the mainstem has the capacity to transport [259]. Small streams are typically regulated by flood-control dams that both trap sediment and reduce the magnitude of high-flow events; the effect on sediment transport is therefore variable and unpredictable [260]. Several reviews have shown that logging and road building increase sediment yield [17, 261–263], primarily via increases in sediment availability due to soil disturbance [102], bank destabilization [264] or landslide acceleration [106, 265]. However, in some cases, post-harvest increases in sediment export may be due to increases in streamflow that result from reduced evapotranspiration rates and interception losses; high channel flows in turn activate in-channel sources of sediment [110, 111]. Others have further proposed that higher water tables throughout the watershed also increase hydrologic connectivity by activating zero-order basins and ephemeral reaches, resulting in more frequent sediment delivery to perennial channels [109]. Only a few studies have examined the effects of harvesting practices on streambed infiltration in small streams, and with variable results; one found significant increases in bed sediment following logging [266], but others found only road construction increased infiltration [267] and only at low flows [268].

4 Biological significance

Fine particles play a major role in stream ecology, but their specific impact depends on particle composition and stream characteristics, including discharge level, bed and channel morphology, and invertebrate-community composition, among others. Organic and inorganic particles may have distinct effects, due to differences in density and nutritional quality. Particles stored in the bed will impact benthic communities, while those suspended in the water column will affect free-swimming consumers and water quality. We begin by reviewing the biological impact of organic and inorganic particle infiltration and then discuss the effect of suspended particles.
4.1 Impacts of fine particle infiltration into the streambed and hyporheic zone

Numerous studies have documented the deleterious impact of increased fine particle deposition to benthic habitats, the hyporheic zone, and associated organisms. Deposition and infiltration of fines into the hyporheic zone, referred to collectively as “streambed clogging,” modifies substrate conditions, trophic resources, and predator activity with ensuing effects on community structure, including an increase in drift and decline in abundance, followed by higher mortality and lower productivity [186]. Progressively, burrowing and fines-adapted assemblages replace invertebrates that require interstitial spaces for habitat [186, 269, 270]. Several studies have shown that fine sediment deposition will decrease macroinvertebrate diversity and abundance [3, 271–273]. An increase in egg and larval mortality of fish species is also commonly linked to the degradation of spawning habitat by fine particle deposition and infiltration into gravel- and cobble-bed sediments [274–276]. In addition, homogenization of the substrate by fine sediment deposition reduces the productivity of algae and the respiration of benthic biofilm [277]. In contrast, the increase in particulate organic matter from decaying post-spawning salmon in western North America has been shown to enhance the growth of stream bacteria and algae [278, 279]. Zones of FPOM accumulation correspond to high microbial activity and degradation rates. The amount of organic matter produced in-channel and biological activity is directly related to the efficiency of particle retention [117, 280].

In addition to direct effects on benthic organisms, fine particle infiltration also blocks intergravel flow and restricts the exchange of oxygen, water, and nutrients vital to benthic organisms [180, 186, 281, 282]. Oxygen supply may be restricted by a layer of sand at the surface of the streambed reducing surface–subsurface exchange [239] or by fines within the gravel pore space that reduce interstitial flow volume and oxygen delivery. Both inorganic and organic particles can reduce oxygen supply via these two mechanisms, but oxygen levels are further reduced by the metabolism of organic particles by bacterial communities [283]. How much organic matter infiltrates into gravel spaces depends on the type and amount of organic inputs as well as the degree of flocculation; flocculation with mineral substances serves to enhance the settling and storage of low-density organic particles [284, 285]. In turn, flocculated fine particle size and density changes according to season and the type of organic matter source; studies from salmon streams of British Columbia demonstrate that the largest, least dense particles are associated with salmon carcasses and die-off periods [285, 286].

4.2 Impacts of suspended particles

High concentrations of particles suspended in the water column can harm the feeding habits of free-swimming consumers such as filter-feeding invertebrates and fish [287, 288] and degrade water quality. Increases in fine particle loads may lead to higher turbidity, as well as eutrophication and high toxicity of both the stream
and receiving water bodies. Water quality is typically assessed according to concentrations of suspended particles and dissolved solutes, including nitrogen (N), phosphorous (P), organic carbon (DOC), and major cations and anions, and occasionally turbidity, conductivity, black disk visibility, pH, alkalinity, or temperature [289–291]. Fine particles, especially those <63 micrometers and those with high organic content, are highly electronegative and therefore strongly associated with the sorption and transport of hydrophobic pollutants and nutrients, including polychlorinated biphenyls (PCBs), dioxins, radionuclides, heavy and trace metals, and nutrients such as N and P [8, 183–185, 292–94].

4.3 Impacts of anthropogenic changes to particle dynamics

Numerous studies have documented a decline in water and bed sediment quality, with ensuing effects on fish and invertebrates, following land-use changes – including agricultural activity, wildfire, dam building, and forest harvesting practices [291, 295–299]. A review of forest harvesting impacts on streams in North America is provided by Binkley and Brown [99]. Typically, any form of vegetation removal and fertilization will degrade water quality by increasing nitrate and suspended-particle concentrations, but consequences vary widely. When intact, floodplain forests act as a filter for pollutants in surface runoff to streams by biological and physical adsorption [300]. High sediment loads and subsequent degradation of spawning habitat following land-use changes has been documented by many researchers [274, 276, 301]. Furthermore, some of these practices (i.e. forest harvesting and road construction) have been linked to an increase in the organic matter content of bed sediments [302, 303], likely due to the erosion of soil organic matter or the delivery of dead plant material from the riparian zone. In some cases, high levels of bed organic matter may not be associated with high inorganic particle yields but may still result in oxygen depletion (see Owens et al. [16] for review).

Some ecological effects may be unique to certain regions; for example, in western North America, logged forests are recolonized by alders, which fix nitrogen and thus produce high-quality, low C:N ratio litter. Stream productivity may increase as a result of this high quality input [304] and higher-quality FPOM, also characteristic of most young, regenerating forests [11]. Higher productivity may also be due to increased decay rates, a consequence of easily decomposed deciduous leaves and an increase in stream temperatures due to the removal of canopy cover and increased sunlight penetration [305–307].

Changes to particle dynamics due to flow regulation can also have profound ecological effects, degrading benthic and floodplain habitat, altering the community composition of aquatic organisms, and decreasing productivity [308–310]. Depending on the style and type of management, dams can also alter thermal regimes or resource availability, with subsequent effects on downstream biotic communities [311] and thus organic matter production. Small streams are typically regulated by dams built for powering mills or flood control, many of which maintain a permanent reservoir. In addition to indirect geomorphic effects on ecology, these dams and associated reservoirs may also directly alter the quantity and
nutritional quality of organic particles and the organisms that feed on them. Reservoir effects at dam outlets, including altered water temperatures and increased FPOM availability, have been well-documented; these are typically linked to an increase in filter-feeding invertebrates downstream [312]. An increase in organic particles below outlets is due to releases from the surface of the reservoir containing lake-derived plankton, otherwise known as seston. Seston quantity and quality then decline with downstream distance, due to factors such as selective depletion by these filter-feeding consumers [313, 314] or dilution by low-quality particles [313, 315].

5 Variability at different spatial and temporal scales

All elements of fine particle dynamics vary across a range of spatial and temporal scales. As a result, inconsistencies between or even within studies may arise from scale-related differences in process or sampling design. Spatial scales range from individual particles to watershed to geographic region. Temporal variations occur on within-year, seasonal scales or over multi-decadal periods of climate and hydrologic change. Nevertheless, relatively few studies have considered the effect of scale on the variability of fine particle dynamics in streams and even fewer have addressed temporal and spatial variation concurrently.

5.1 Spatial scales and variability

Sampling limitations typically restrict most studies to the reach or channel scale. Few measurement techniques operate at a fine enough scale for the monitoring of individual particle movement, while watershed- or larger-scale sampling may be prohibitively expensive and time-consuming. Most studies are confined to a single spatial scale and stream size, although some have attempted to study fine particle dynamics at multiple scales and over large geographic areas. Water quality classification schemes, for example, have been developed to extrapolate data collected at a few sites to an entire region or to subdivide large areas into smaller zones (see review by Roberston and Saad [316]). Three broad types of extrapolation/classification schemes exist, including those based on similar environmental characteristics, regression equations, and mechanistic models, as well as a combination of these three. However, variability in these classification schemes arises due to uncertainties in data representativeness, within-zone variation, and the selected environmental factor or water-quality metric (e.g. total N, P, or DOC). A recently developed approach, known as SPARTA, attempts to avoid these problems by using a regression-tree analysis and GIS coverage to subdivide an area into homogenous zones specific to a given water-quality metric and based on the environmental factor most statistically important to that metric [316].

Management and restoration of degraded watersheds necessitates an understanding of which factors, and which scales, have the greatest effect on stream conditions, yet debate remains over which spatial scale contains the best indicators.
of water quality and in-channel ecological conditions. In recent years, advancements in geographic information systems (GIS) techniques have allowed researchers to determine landscape metrics over large areas and at fine scales [317]. Combined with in-channel measurements, spatial land-cover and land-use databases can be used to determine which variables best explain stream conditions. Several studies indicate that watershed or landscape factors, such as soil type, geology or land use, are better predictors than local stream characteristics, such as riparian condition [181, 317–319], but the reverse has also been found [320]. A combination of effects at both the local and watershed scale may also explain in-channel biological conditions [321, 322]. Landscape metrics can be successful predictors of water quality, as shown by Jones et al. [323] who find that metrics generated from easily obtained spatial data, such as agricultural land cover, extent of riparian forests, and amount of atmospheric nitrate deposition explain most of the variability in dissolved N, P, and suspended sediment. Most studies addressing this issue, however, have been limited to only one or a few adjacent watersheds with similar climate, land use, and fish species. In contrast, Meador and Goldstein [290] use data from 20 major U.S. river basins and a multimetric approach to assess the relationship between land use, riparian condition, water physicochemistry, and fish community structure condition over a broad regional geographic scale. Across large geographic scales, they suggest that water physicochemistry indices are better indicators of fish community condition than watershed land use. Results also indicate that across all the regions studied, riparian condition and associated practices regulate the delivery of nutrients, suspended particles, and dissolved solids to streams, in turn affecting the fish community condition. Despite their relative success, predictive models used in these types of studies contain a certain degree of spatial variability that is inherent to the processes of particle supply and transport.

5.2 Temporal scales, trends, and variability

Temporal variability adds another dimension of complexity that can be exacerbated by ill-suited sampling schemes and techniques. As a result, researchers have suggested that sampling frequency and technique be selected according to nutrient status [324] or statistic of interest and study duration [325]. Long-term monitoring programs typically take manual samples at fixed intervals (i.e. monthly), while short-term studies may be more intensive, taking continuous, automatic samples during flood events and fixed-interval manual samples during supposedly stable intervening flows. From these samples, various statistics, such as annual load or maximum concentration, and rating relations may be calculated to assess water quality. A review and analysis of the cost-effectiveness, precision, and accuracy of approaches for monitoring water quality in small streams is provided by Robertson and Richards [316].

For a long time, researchers have recognized that water quality and discharge characteristics vary seasonally [326, 327], but recent studies have also documented large fluctuations during rain events [328, 329]. Typically, seasonal variability is
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controlled by precipitation patterns and factors related to land cover, such as evapotranspiration, interception, and infiltration [330]. During rain events, rising discharge can either increase suspended particles via entrainment from the bed or increased surface runoff [331] or dilute cations and suspended biota if little excess is supplied from the watershed [332–334]. Biological retention and physical sorption processes may also decline with increased discharge levels [335, 336].

Hydrologic variability typically changes with basin size; groundwater inputs to small streams and high water volume in large rivers dampen temporal fluctuations in discharge and water quality, such that mid-sized streams are assumed to have the greatest environmental variability [34], with implications for biological productivity and diversity [34, 337]. Despite the influence of river size on hydrologic variability, however, a recent study by Chatellet and Pick [324] demonstrates that at regional and watershed scales, the seasonal variability of water-quality characteristics does not change with river size. Instead, temporal variability and the requisite sampling frequency depend on the water-quality metric, the time of year, and in some cases, the mean concentration. Other spatial characteristics may also influence temporal variability, such as spatial patterns of vegetative cover, subsurface flows, and soil moisture, which control rates of surface erosion and runoff to the stream [186, 330].

Although intra-annual variability has been relatively well studied, documentation of long-term temporal changes in hydrology and associated sediment and solute dynamics is limited by a paucity of long-term data sets. Nevertheless, several methods, including the analysis of paleoenvironmental archives (i.e. archaeological remains, tree rings, documents and instrumental records), numerical simulations (i.e. runoff and erosion models), and long-term monitoring programs, allow researchers to elucidate temporal trends in discharge, particle, and solute dynamics over sub-decadal [338–341] to millennial [342, 343] time scales, often focusing on the impact of climate change and human activities. Paleoenvironmental studies and long-term monitoring programs provide information on past conditions that can facilitate understanding of present-day systems and future trajectories, as well as provide parameterizations for numerical simulation models [344, 345]. Simulation models are typically used to reconstruct or predict how changes in environmental variables, such as climate or land disturbance, affect particle and solute dynamics [346, 347].

Over the past century, regions throughout the world have experienced severe changes in land use and climate. Although anthropogenically induced climatic warming over the past century has had a significant impact on hydrologic and sediment regimes in systems throughout the world [345, 348, 349], climate change often acts concurrently with human activities such as water diversion and consumption [341, 345] or land disturbance [350–352]. For this reason, determining the relative impacts of climatic versus land-use changes can be difficult, but is often attempted through multiple regression models [341] or spatial and chronological links [340, 350, 351].

Results from small streams suggest that land-use changes and local catchment conditions have a dominant effect on long-term temporal trends of sediment
transport and water quality, reflecting the strong link between stream and hillslope processes [338]. For example, despite increases in precipitation, reforestation has been shown to reduce soil erosion rates and overall sediment yield [350, 351]. In contrast, sediment cores from headwater lakes in several UK catchments chronologically link high sedimentation rates to land drainage, land conversion, and afforestation, but not to temporal climate changes [340]. Non-climate-related human impacts are not unique to small streams, however. Walling and Fang [345] assembled long-term records of annual sediment load and runoff for 145 major rivers and found highly variable, mostly insignificant temporal trends. Evidence from this and other studies indicates that reservoir construction and the consumption or diversion of water by humans, may be the most important influence on large river sediment fluxes [341].

Several studies demonstrate that the magnitude, direction, and mechanism of change will depend on catchment factors such as river size [338, 352], local climate [352–354], or underlying geology [348, 349]. Decreasing sediment yields contradict climate models that predict increasing global precipitation and temperatures will subsequently increase erosion and runoff rates [339, 350], perhaps reflecting either unchanging controlling variables or buffering mechanisms within the watershed [345, 354]. Increasing sediment yields is reported in several areas with intense human activity, such as deforestation, agriculture, and road building, which increase erosion rates [352, 355, 356], but climate appears to play only a secondary role. Relatively consistent sediment yields [357, 358], despite significant climatic and environmental changes, may be due to alluvial storage and remobilization, which keeps the fluvial system in a state of dynamic stability. It is proposed that any external changes are buffered by internal reorganizations in sediment deposition and negative feedbacks on weathering rates [359].

Natural climatic fluctuations occurring on interannual to millennial timescales are shown to influence the temporal variability of sediment dynamics and geomorphic processes [360]. Several studies document the impacts of climate changes induced by sub-decadal La Nina and El Nino/Southern Oscillations and Pacific Decadal Oscillations [348, 349, 361–363], but results vary between regions. Strong El Nino events produce significant increases in sediment yield in some areas [349, 362] and decreases in others [348, 363, 364]. Events can also alter the composition of sediment loads, including the concentration of particulate nutrients, metals, and organic matter, with effects on productivity and contaminant concentrations of receiving waters [349, 361, 364, 365]. In several cases, factors such as basin geology [348, 349], vegetative cover [364], or impoundment [363], influence the degree of climatic effect by controlling erosion rates or water fluxes.

As revealed in the discussions above, spatial and temporal effects are closely linked; both must be incorporated into a study design. Selecting the appropriate scale for study of a given system requires understanding which factors and processes dominate the subject of interest and the degree of variability inherent to each. Factors such as human activities, geology, location, vegetation, river size, and climate can influence the response of the system and should be considered when selecting measurement techniques, sampling design, and data analysis.
6 Research needs

One major research need in the field of fine particle dynamics are the factors influencing the vertical movement and exchange of fine particles, particularly streambed deposition and infiltration. As demonstrated in this chapter, a complex combination of biological and physical factors will influence the distribution, deposition, and infiltration of particles. A major remaining question is why measured rates of particle deposition often differ markedly from particle settling velocities; although several mechanisms have been proposed to explain this discrepancy, it is not fully understood. Still largely unexplored are the roles of transient storage zones, biological controls such as particle trapping by biofilm or removal by invertebrates, and particle composition. Also largely unexplored is the relative importance of particle flocculation and organic matter content on particle transport and deposition. Recent studies suggest that flocculation may affect near-bed concentrations [366], bed-sedimentation rates [367, 368], and metal adsorption [369]. Organic matter content will also affect biological consequences; higher nutritional quality may increase benthic productivity, but decomposition may lead to more pronounced oxygen depletion within the hyporheic zone. Thus, both the physical and biological impacts of flocculation are still uncertain. In addition, only a few studies have measured the response of streambed deposition and infiltration rates to changes in sediment yield, organic matter inputs, and water discharge following forest harvesting.

A second major area in which more research should focus is the development of more accurate and representative models of vertical exchange and distribution. As a consequence of our limited understanding and the complexity of fine particle dynamics, current models of vertical exchange and distribution are constrained by assumptions and conditions unrealistic for most natural systems. In particular, no available model provides an explicit representation of bed composition or bed complexity, particle composition, or benthic ecology. Traditional hydrodynamic models of vertical particle distribution (Rouse equation, advection-dispersion model) are limited because they do not consider the small-scale mechanics of particle motion, particularly the effect of particle–water interactions on turbulent flows and particle movement. The Local Exchange Model improves upon these models by describing the movement of an individual particle due to turbulence as a stochastic diffusion process that includes a depth-dependent measure of vertical dispersion. Nevertheless, this model is still limited because it does not consider the effect of particles or the effect of complex bed topography on turbulent flows. An alternative approach to modeling the vertical movement of suspended particles has recently been presented by Muste et al. [114] in which particle and water movement are treated as two distinct phases rather than as a single mixed fluid. However, no theoretical model exists that can fully encapsulate the numerous factors influencing vertical particle exchange. A conceptual model of particle exchange that includes physical and biological mechanisms is needed as a framework on which to base our predictions. As a starting point to developing this model, we need more accurate measurements of the vertical distribution of particles under a range of hydrological, geomorphic, and biological conditions.
A third major research need is the development and refinement of instruments and measurement techniques for particle movement and infiltration. Newly introduced devices such as the infiltration cube require further testing for different conditions and study objectives. In addition, the use of radionuclides for tracking particle movement is a promising new field that should be explored. Short-lived radionuclides have been used to quantify the longitudinal transport of fine sediment, but may also serve as tracers in the vertical direction, indicating rates of deposition, depths of infiltration, and degree of streambed mixing. Short- and long-lived radionuclides could be used in combination to identify particle residence times in both long-term and transient storage zones, as well as spatial variability in particle storage across the stream channel.

Clearly, despite extensive study on the dynamics of fine inorganic and organic particles in streams, gaps in our knowledge still remain. Many complex and interrelated factors govern the movement, storage, and impact of fine particles, factors that vary greatly over spatial and temporal scales. Human activities can also profoundly alter and further complicate these processes. Generally, the most significant limitation to our understanding of fine particle dynamics is the predominantly one-sided focus of most studies. Increasingly, researchers are recognizing the important links between physical and biological components, but truly cross-disciplinary studies are rare. One of the most important advancements in coming years will be the successful integration of the approaches, methods, and knowledge of these two fields.

References


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