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Erosion processes in steep terrain—Truths, myths, and uncertainties related to forest management in Southeast Asia

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Abstract

To assess the effects of forest management on soil erosion in Southeast Asia, clear distinctions must be made between surface erosion and landslide processes. Although surface erosion is a natural process, it is exacerbated by surface disturbance and compaction that reduce the soil hydraulic conductivity and break down soil aggregates. Management practices and attributes such as roads and trails, agricultural cultivation, fire, land clearing, and recreation all accelerate surface erosion processes due to their disturbance, compaction, and connectivity along hillslopes. Agroforestry practices in Southeast Asia that incorporate cover crops with trees reduce surface erosion by more than an order of magnitude compared to monoculture plantations with no ground cover. Cleared fields tilled up and down steep slopes are highly erodible; passive conservation practices (e.g., contour tillage, strip cropping, reduced tillage; maintaining adequate ground cover) are effective in reducing surface erosion if properly implemented. Poorly designed and managed terraces are not effective in controlling surface erosion and may actually increase mass wasting if they concentrate water.

In contrast to surface erosion, shallow, rapid landslides are episodic processes triggered by individual rainfall events or artificial inputs of water; slower, deep-seated landslides initiate or activate after a longer-term accumulation of water. Thus, landslide assessment must be based on long-term observations. Deep rooted trees and shrubs impart a significant cohesive strength into shallow soil mantles and facilitate preferential drainage, thereby reducing the probability of shallow landslides. Conversion of mountain forests to cropland or plantations permanently reduces rooting strength, thus increasing landslide potential, while timber harvesting with subsequent regeneration of secondary forests reduces rooting strength for up to two decades after initial cutting.

Roads contribute the largest surface erosion and landslide losses (per unit area disturbed) compared to other land uses. Both landslide and surface erosion fluxes along roads are typically one to more than two orders of magnitude higher compared to undisturbed steepland forests. High storm runoff from roads is caused by the generation of infiltration-excess overland flow on compacted surfaces and the interception of subsurface flow at road cuts; these altered pathways increase surface erosion and accelerate the delivery of storm runoff to streams. Discharge nodes from roads facilitate the connectivity of water and sediment to headwater streams. Trails, although narrower than roads, can contribute significantly to soil loss and storm runoff and are important sediment conduits if directly linked to channels. Thus, the location of roads and paths with respect to the hydrologic network is a critical factor governing the spatial–temporal movement of sediment and water in the tropics.

Keywords: Tropical forest management; Landslides; Surface erosion; Forest roads and paths; Forest conversion; Hydrologic pathways; Tropical agricultural; Agroforestry

1. Introduction

The sustainability of recent upland management practices in Southeast Asia is a topic of intense scrutiny and concern, but few practical solutions have emerged to address this issue (e.g., Bryant and Parnwell, 1996; Craswell et al., 1998; Laurance, 1999; Cramb et al., 2000; Tomich et al., 2004). Government and international donor incentives that promote forest conversion to high cash value crop production together with poorly coordinated conservation programs, transmigration schemes, ineffective participatory approaches to catchment management,

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rising market prices for certain crops, and the apparent needs of subsistence farmers to generate additional sources of short-term income from the land have placed increasing demands and pressures on soil and water resources (e.g., Byron and Arnold, 1999; Elmhirst, 1999; Cramb et al., 2000; Lu et al., 2001; McMorrow and Talip, 2001; Thapa, 2001; D'haeze et al., 2005). Such impacts are most severe in mountainous terrain where hydrologic processes are altered by land disturbances and compaction causing accelerated surface erosion and episodic landsliding, and subsequently increasing sediment delivery to streams (e.g., Douglas et al., 1999; Chappell et al., 2004; Sidle et al., 2004a; van Dijk and Bruijnzeel, 2004; Ziegler et al., 2004a). Herein, these two processes (surface erosion and landslides) are distinguished in terms of their causes and impacts related to forest practices and conversion in tropical Southeast Asia.

Sustainable management of upland forests and agricultural lands is difficult to accomplish in Southeast Asia because the combination of steep hillslopes, high rainfall intensities, seasonally dry periods, and naturally erodible and unstable soils promote surface erosion and landslides (Sidle et al., 1985, 2004a; Chappell et al., 1999; Douglas et al., 1999; Lu et al., 2001; Chang and Slaymaker, 2002). Land uses that contribute to the majority of the surface erosion and slope stability problems in tropical mountain uplands include: timber harvesting, roads and trails, various agroforestry practices; conversion of forest to agricultural land; and grazing (e.g., Sidle et al., 1985, 2004a,b; Hashim et al., 1995; Lu et al., 2001; Bruijnzeel, 2004; Ziegler et al., 2004a,b). Recreation appears to be having an increasing impact on erosion, including small-scale mass wasting, but the effects are poorly documented and typically occur as the result of trails and off-road vehicle use (Sutherland et al., 2001; Sidle and Dhakal, 2002). The conversion of tropical forests to commercial and non-commercial agricultural lands can have significant effects on water pathways and soil erosion, thus influencing site productivity (e.g., Harwood, 1996; Ross and Dykes, 1996).

The objectives of this paper are to illustrate how the different erosion processes are influenced by site and dynamic hydrologic conditions and to demonstrate (via original data and simulations, as well as previously published field examples) how timber harvesting, forest conversion (to agriculture), and roads and trails potentially affect both surface erosion and landslide processes. These 'truths' associated with interactions of forest management and erosion processes are contrasted to some of the 'myths' perpetrated in the literature or in generalizations related to land management in the region. Additionally, gaps in knowledge ('uncertainties') related to erosion processes and fluxes, as well as forest management and conversion effects in Southeast Asia are elucidated.

2. Impacts of surface erosion and landslides

The productivity of forests and hillslope agricultural lands can be significantly reduced due to accelerated erosion processes, both surface erosion and mass wasting. Much of the nutrient capital of these soils is associated with the organic-rich topsoil, which is subject to displacement via surface erosion following disturbances (Ross and Dykes, 1996; Rai and Sharma, 1998; Fu et al., 2000). The disturbance and erosion of topsoil decreases the available water in residual soils; these decreases limit vegetation growth, especially in seasonally dry sites (Andreu et al., 1998).

Landslides impact site productivity more severely but usually over much smaller areas compared with surface erosion (Fig. 1a). Entire soil mantles are often stripped away leaving only the parent material exposed (Crozier et al., 1980; Trustrum et al., 1983). Recovery of site productivity on landslide scars appears to be influenced by size of the scour zone, infilling rates of soil and organic matter into the previously failed area, availability of seed sources, nature and weathering characteristics of substrate, colonization by nitrogen-fixing vegetation, available nutrients, and soil water conditions (Harris, 1967; Okunishi and Iida, 1981; Adams and Sidle, 1987; Shimokawa et al., 1989), together with subsequent geomorphic disturbances (Douglas et al., 1999). Studies on the recovery of landslide sites in the tropics are lacking, but high rates of rainfall and weathering should promote more rapid recovery than in temperate zones (Shimokawa et al., 1989; Douglas et al., 1999).

Long-term increases in surface erosion and landslide sediment exert off-site impacts, including downstream channel changes and sedimentation (e.g., Rust, 1972; Xu, 1991; Douglas et al., 1999), deteriorated water quality (e.g., El-Swaify, 1987; Chappell et al., 2004), and damage to downstream fish populations and aquatic habitat (e.g., Lambereti et al., 1991; Martin-Smith, 1998). Concentrated overland flow can create gullies that act as efficient corridors of sediment transport to streams and cause channel head cutting (Croke and Mockler, 2001; Poesen et al., 2003; Sidle et al., 2004a,b) (Fig. 1b). Although landslides are naturally occurring phenomenon in steep forested terrain, they pose significant hazards to humans and property (e.g., Sidle et al., 1985; Swanston and Schuster, 1989; Lu et al., 2001; Sidle and Chigira, 2004).

3. Contrasts between surface erosion and landslide processes

Conceptually, the difference between surface erosion and landslide processes is quite clear—surface erosion is a water-driven process, whereas landslides are gravity driven. Nevertheless, surface erosion is influenced by gravity (i.e., slope gradient affects runoff velocity and thus surface erosion potential) and landslides are affected by water (i.e., pore water pressure reduces soil shear strength). However, confusion arises in the interpretations of the influence of hydrologic events and inputs on these processes with respect to upland catchment management (e.g., Baharuddin and Abdul Rahim, 1994; Thomas, 1994; Douglas et al., 1999; Rasul and Thapa, 2003; Bruijnzeel, 2004). A clear understanding of the linkages between hydrologic pathways and erosion mechanisms is critical to evaluating the effects of forest land uses on both surface erosion and landslides. A complete synthesis of these effects is lacking for mountainous areas of tropical Southeast Asia.

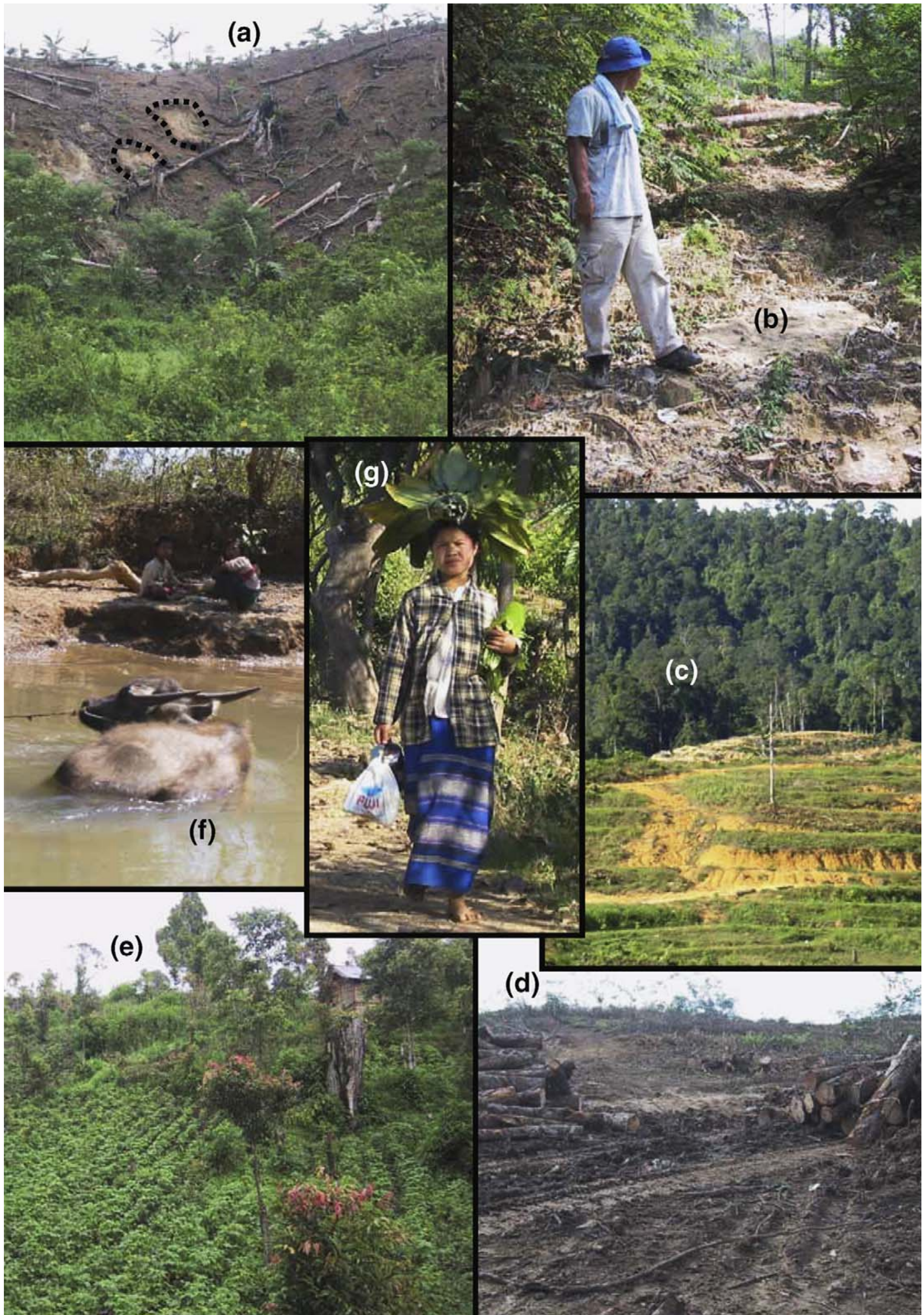

Fig. 1. (a) Formerly forested hillslope in Sumber Jaya, Sumatra, Indonesia, that has been converted to a monoculture coffee plantation; virtually no ground cover exists and significant surface wash has occurred together with small, shallow landslides (highlighted with dashed lines); (b) storm runoff from logging roads created gullies on hillslopes and caused channel headcutting, Peninsular Malaysia; (c) extensive surface erosion and small-scale mass wasting on terraces constructed for oil palm plantations near Terengganu, Malaysia (photo by Shozo Sasaki); (d) extensive disturbance and soil erosion following forest clearance for the establishment of rubber plantations on slopes in Kerling, Malaysia; (e) relatively sustainable agroforestry practice in the headwaters of Sumatra: cinnamon trees grown with a legume ground cover; (f) extensive compaction and disturbance by water buffalo around a stream draining into Inle Lake, Myanmar; vegetation removal and trampling also caused localized bank failures; (g) ubiquitous footpaths used by villagers in central Myanmar (here transporting vegetables near Inle Lake).

Table 1
Estimated values of saturated hydraulic conductivity (K_{sat}) for selected tropical forest soils in Southeast Asia with different disturbance regimes

Sites	Data sources	Estimated saturated hydraulic conductivity, K_{sat} (mm h^{-1})		
		Undisturbed forest	Roads/skid trails	Other disturbances
Cikumutuk Catchment, West Java, Indonesia	van Dijk (2002)			260–426 (terrace beds)
Jengka Experimental Basin, Peninsular Malaysia	Baharuddin et al. (1996)	33[a]	23–24[a]	
Bukit Tarek Experimental Watershed, Peninsular Malaysia	Noguchi et al. (1997a)	1466 (10 cm depth), 790 (20 cm depth), 585 (40 cm depth), 169 (80 cm depth)		
Bukit Tarek Experimental Watershed, Peninsular Malaysia	Negishi et al. (2004, 2006)	203–2214; mean: 674 (10 cm depth) >0.1–33; mean: 2 (75 cm depth)	0.026	
Sipitang, Sabah, Malaysia	Malmer (1996a)	Clay topsoil: 0.68 (20 cm), 0.16 (40 cm); sandy topsoil: 0.11 (20 cm), 0.033 (40 cm)	Clay topsoil: 0.032 (20 cm), 0.013 (40 cm); sandy topsoil: 0.176 (20 cm), 0.052 (40 cm)	
Upper Segama area, Sabah Malaysia	van der Plas and Bruijnzeel (1993)		0.5–45	
San Mun, northern Thailand	Ziegler and Giambelluca (1997), Ziegler et al. (2000)	146–524	0.2–5.1; mean: 2.3; 8–15[b] (mean values)	14–105[c]; 129-316[d]
Tanh Minh, North Vietnam	Ziegler et al. (2004a)	12–290; mean: 91	2–43; mean: 11	27–112[e]

[a] Mean values for slopes ranging from 10 to 30%; values are derived from infiltration 'rates', thus K_{sat} could differ.
[b] Mean values for unpaved, secondary roads and compacted paths.
[c] Mean values for agriculture, secondary vegetation, and roadside margins.
[d] Mean values for field paths and upland, hoed and fallow fields.
[e] Mean values for upland fields, abandoned fields, young secondary vegetation, intermediate secondary vegetation, and grasslands.

3.1. Surface erosion

In most natural and managed forests, the infiltration capacity and hydraulic conductivity of surface soils are relatively high (Table 1). High infiltration capacities are supported by continual inputs of organic matter onto the soil surface. Because tropical forest soils experience high rates of decomposition, organic horizons are thin compared to temperate soils (Brown et al., 1994; Hairiah, 1999; Lal, 2002). In undisturbed forests, precipitation generally infiltrates into the soil and moves to streams as subsurface flow (e.g., Sidle et al., 1995; Noguchi et al., 1997b; Douglas et al., 1999; Bruijnzeel, 2004). Exceptions may occur in sites with a low permeability layer near the surface (e.g., Bonell, 1993; Elsenbeer and Vertessy, 2000) that promotes return flow during storms with wet antecedent conditions. However, many tropical soils exhibit marked decreases in hydraulic conductivity in the upper portion of the soil profile and still transmit most water to streams via subsurface flow (e.g., Bidin et al., 1993; Elsenbeer and Lack, 1996; Malmer, 1996a; Noguchi et al., 1997b). Because subsurface flow predominates, surface erosion is usually low in undisturbed tropical forest catchments (e.g., Malmer, 1996a; Bruijnzeel, 2004). Riparian corridors, geomorphic hollows, and areas with thin soil cover over low permeability substrate typically experience saturated overland flow during storms (Dunne, 1983; Chandler and Walter, 1998; Sidle et al., 2000). Many sites of saturated overland flow are either too flat or spatially isolated for this pathway to initiate significant surface erosion.

Raindrop impact on exposed soil surfaces can detach and displace soil particles causing surface sealing or crusting, and thereby initiating surface erosion on upper portions of slopes (e.g., Le Bissonnais, 1990; Morgan, 1995; Watung et al., 1996). The short-duration, high-intensity monsoon storms that are common throughout much of Southeast Asia provide the necessary rainfall erosivity to drive this process (Free, 1960; Quansah, 1981; Douglas et al., 1999). Multi-tiered tropical forest canopies and ground cover protect mineral soils from splash erosion (e.g., Hashim, 1988; Rostagno, 1989; Bruijnzeel, 2004); however, when canopy height is uniform, raindrop diameter typically increases (e.g., Nanko et al., 2004). Significant surface erosion is generated by overland flow, especially when flow channelizes, forming rills and gullies (Poesen et al., 2003). The thin litter layer of many tropical soils is easily displaced by overland flow, thus exposing mineral soils to erosive processes (Hashim, 1988; Hartanto et al., 2003; Sidle et al., 2004b). Other site attributes including slope gradient, topographic complexity, aggregate stability, exposed rocks, root mats, and woody debris on the ground surface affect surface erosion potential (Poesen, 1985; Hashim, 1988; Ross and Dykes, 1996; Chappell et al., 1999; Hartanto et al., 2003; Ziegler et al., 2004a).

3.2. Landslide processes

Shallow subsurface flow, particularly during rainstorms, accumulates in various hillslope positions due to topography (surface and subsurface) and continuity of soil hydrologic pathways (Anderson and Kneale, 1982; Fernandes et al., 1994; Tsuboyama et al., 2000). If combined with steep slopes and soils that overlie relatively impermeable substrate, high pore water pressures can develop and trigger a landslide. Such shallow, rapid landslides naturally occur in mountainous forest terrain, particularly in geomorphic hollows (also called zero-order basins) (Tsukamoto and Ohta, 1988; Sidle and Wu, 1999). Little can be done to prevent such landslides and they are major contributors of 'natural' episodic sediment to streams. A common hydrologic sequence for shallow landslide initiation involves wet antecedent conditions followed by a prolonged period of rainfall with a burst of high intensity (e.g., Sidle and Swanston, 1982). Deep-seated landslides, such as slumps and earthflows, generally initiate and accelerate in response to a gradual buildup of pore water pressure over weeks or even an entire rainy season (e.g., Wasson and Hall, 1982; Iverson and Major, 1987). Water recharge into deeper soils is controlled not only by 'small-scale' hydraulic properties (i.e., hydraulic conductivity changes with depth), but also by 'larger scale' phenomena, such as surface tension cracks and fractures in bedrock (Sidle et al., 1985; Montgomery et al., 1997; Sidle and Chigira, 2004). For both shallow and deep-seated landslides, disruption of subsurface flow and flow characteristics in the regolith influence pore water pressure response, and thus, the potential for landslide initiation. The major risk to humans occurs with shallow, rapid landslides and the occasional deep, rapid landslides that impact areas where humans settle; conversely, slower, deep-seated landslides rarely cause loss of life, but can inflict extensive property and environmental damage.

In steep terrain, forests protect against landsliding in several ways: (1) modifying soil moisture regime through evapotranspiration; (2) providing root cohesion to the soil mantle; (3) maintaining secondary permeability in the soil. High year-round evaporation rates of tropical forests (Greenway, 1987; Bruijnzeel, 2004) act to stabilize hillslopes by preventing pore water pressure from exceeding critical thresholds that will trigger landslides. Loss of soil water via evapotranspiration by trees can also limit the period of shallow landslide susceptibility as well as the period of deep-seated landslide activity. Deep-rooted vegetation species growing in deep soils sustain high transpiration rates for long periods, thus drying soils at greater depths compared to shallow-rooted vegetation (McNaughton and Jarvis, 1983), and decreasing landslide potential. Tree roots (both alive and dead) contribute to macropore formation in tropical soils, and together with faunal activity and buried pockets of decaying wood, form preferential flow networks that often provide effective lateral drainage of hillslopes (e.g., Noguchi et al., 1997a; Sidle et al., 2001). Such drainage paths could be very important when soils approach saturation to dissipate the formation of zones of positive pore water pressure that could potentially trigger landslides. Chandler and Bisogni (1999) used tracer investigations to infer extensive alterations of subsurface flow pathways due to conversion of tropical forests to pasture and tilled agricultural fields in Leyte, Philippines. The implications are that tropical catchment disturbances disrupt preferential flow paths thereby creating possible (but unproven) scenarios for pore water pressure accretion and landslide initiation.

The contribution of vegetation roots to soil shear strength is generally believed to be greater than either the evapotranspiration or secondary permeability effects (Wu et al., 1979; Gray and Megahan, 1981; Abe and Ziemer, 1991). Shallow soils are much more influenced by rooting strength than deeper soil mantles. In shallow soils, roots may penetrate the entire soil mantle, providing vertical anchors into more stable substrate (Wu et al., 1979; Gray and Megahan, 1981; Greenway, 1987). Dense lateral root systems in the upper soil horizons form a membrane that stabilizes the soil (Sidle et al., 1985; Schmidt et al., 2001). This membrane is much more significant in protecting against shallow landslides than deep-seated landslides (Swanston and Swanson, 1976). Tree roots may lend some stability to deeper soils by lateral reinforcement across planes of weakness (Swanson and Swanston, 1977; Schroeder, 1985); however, this beneficial effect would diminish with larger and deeper potential failure sites (Burroughs et al., 1985).

The timing of sediment entry and transport through streams is affected by the initial travel distance of the landslide (and whether it mobilizes into a debris flow), the extent of large wood in the channel, tributary junction angles, and channel and hillslope gradients (Benda and Cundy, 1990; Douglas et al., 1999; Gomi et al., 2002, 2006). As a result, sediment export through disturbed headwater areas (e.g., roads, harvesting, fire) may be delayed by several years (Douglas et al., 1999; Malmer, 1996b, 2004). In contrast, surface erosion may be routed more efficiently because of their smaller particle size (Sidle et al., 2004b; Gomi et al., 2006).

4. Implications for forest management activities in the tropics

This section focuses on interactions between land management activities and erosion processes in tropical Southeast Asia, namely timber harvesting, forest conversion to agriculture or plantations, roads and trails, agroforestry, and grazing. This summary is not comprehensive, but provides insights with illustrative examples of the magnitudes and types of surface erosion and landslide problems, as well as linkages with dynamic site and hydrological conditions associated with typical land uses in Southeast Asia.

4.1. Timber harvesting

4.1.1. Surface erosion

Surface erosion associated with felling and yarding trees in tropical forests is generally quite small unless extensive mineral soil is exposed (Malmer, 1996a,b; Hartanto et al., 2003; Bruijnzeel, 2004) (see Example 1). The extent of surface erosion on forest hillslopes depends on the depth of the disturbance related to the depth of the soil organic horizons, as well as the spatial extent or connectivity of disturbances. Alterations in dynamic rainfall-runoff response (particularly infiltrability) on sites affected by timber harvesting will determine the amount of surface erosion that may occur (Malmer, 1996a; Noguchi et al., 1997b). Since tropical forests usually have thin organic horizons, ground-based yarding systems (e.g., crawler tractors, rubber-tire skidders, cable yarding along the ground perpendicular to the slope contour) may disturb soils, thereby increasing the risk of surface erosion (e.g., Baharuddin et al., 1996; Malmer, 1996a; Douglas et al., 1999). However, some mixing of the organic and mineral soils during harvesting may not significantly increase surface erosion if infiltration capacities of mineral soils remain high (Malmer, 1996a).

A recent study in Central Kalimantan, Indonesia, found no increases in surface erosion on both conventionally harvested and reduced impact logging plots compared with control plots; however, loss on skid trail plots, where compaction and extensive disturbance occurred, was more than three orders of magnitude higher (Hartanto et al., 2003). While significant increases in both suspended and bedload sediment at catchment outlets have been reported after logging of forests in various parts of Malaysia (Baharuddin and Abdul Rahim, 1994; Baharuddin, 1996; Malmer, 1996a,b; Douglas et al., 1999), it is generally acknowledged that harvesting alone does not substantially contribute to these increases; rather logging roads, skid trails and other highly disturbed sites constitute the primary sediment sources (e.g., Burgess, 1971; Baharuddin et al., 1995; Sidle et al., 2004b) (see Example 1). Because most catchment-scale studies do not specifically quantify such chronic sediment sources, reported sediment yields must be carefully assessed in terms of their relationship to land management practices.

4.1.2. Landslides

In contrast to surface erosion, landslide erosion begins to increase several years after timber harvesting. Many field

Box 1. Example 1: Surface erosion—timber harvesting

Findings in managed forests of Sabah, Malaysia suggest that surface erosion losses from harvested plots are quite low (<0.2 t ha^{-1} yr^{-1}; Malmer, 1996a,b). However, surface erosion rates nearly doubled at both the plot and catchment scales when crawler tractors were used for logging compared to manual clearing methods (Table 2). Erosion losses at the catchment scale were about 20-fold higher than at the plot scale in the two managed catchments; however, for the control catchment this increase was about 5-fold. Networks of skid trails and even human trails in the managed catchments contributed to higher erosion losses compared to the plots, while in the control catchments, the more modest increases were likely attributable to scaling effects and other erosion sources (e.g., streambank erosion) (Malmer, 1996b). Mean suspended sediment levels at the outlets of the managed Sabah catchments were five to seven times higher during the most affected time after logging compared to the period before logging (Table 2). In catchment #3, this maximum occurred during tractor yarding but before burning. In catchment #4, where manual felling of secondary vegetation was followed by total biomass burning, peak suspended sediment levels occurred after burning. In contrast, when manual clearing was employed with no burning (#2), the lower maximum suspended sediment levels occurred 1 yr after harvesting. These increases in sediment transport in all managed catchments lasted about 1 yr after the various disturbances, but then returned to control (#1) levels (Malmer, 1990, 1996a,b). About a decade later, the three managed catchments were again harvested; 3–9 months thereafter a wildfire occurred. In contrast to the minor increases in suspended sediment levels in the manually felled catchment (#4), concentrations in the other two logged catchments (#2 and 3) increased by almost an order of magnitude during the 2-yr period after the wildfire (Malmer, 2004; Table 2). This delayed response in sediment transport likely illustrates the sudden release of stored sediment from these two more disturbed catchments, both of which had two to three times the biomass and greater amounts of dead wood compared to catchment #4. It is unclear how much of this source of stored sediment was related to harvesting; however, Malmer (1996a) reported rates of erosion from skid trails ranging from 244–547 t ha^{-1} yr^{-1} in the 2 yr after initial harvesting, and from 77–122 t ha^{-1} yr^{-1}during the third year after logging. This study shows that the type of harvesting and site preparation methods employed (manual clearing alone, residue burning, tractor yarding), affect the magnitude and timing of soil loss. Also, the delayed effects of mobilization of management-induced sediments generated a decade previously were elucidated after wildfire. Thus, short-term monitoring of sediment transport after disturbances may give a distorted picture of forest management influences on sediment dynamics.

Table 2
Annualized surface erosion losses from managed forest plots together with sediment export and suspended sediment concentrations at the catchment scale, Mendolong Research Area, Sabah, Malaysia

Cover/treatment	Plot erosion gradient (15–21°) (t ha^{-1} yr^{-1})	Catchment erosion[a] (t ha^{-1} yr^{-1})	Mean suspended sediment before/after[b] 1987 logging (mg L^{-1})	Mean suspended sediment 10–21 months/22–33 months after 1998 wildfire (mg L^{-1})
1. Control tropical forest	0.038	0.16–0.22	–	
2. Manual clearing of residual forest, no burning	0.11	2.2	4.5/30.5	400–450/5–40
3. Manual felling, extraction of wood by crawler tractors, burning and planting	0.20	3.9	14.5/78.0	
4. Manual felling of secondary vegetation (5 yr after fire); burning all biomass	–	1.1	7.9/59.1	50–70/40

Data from Malmer (1990, 1996a,b, 2004) and Grip et al. (1994).
[a] Based on suspended sediment samples only.
[b] Suspended sediment levels after logging are for the 'most impacted' period measured.

investigations in steep forested terrain worldwide have noted up to a 10-fold increase in landslide erosion in the period roughly 3–15 yr after timber harvesting (Bishop and Stevens, 1964; Endo and Tsuruta, 1969; O'Loughlin and Pearce, 1976; Megahan et al., 1978; Wu and Sidle, 1995; Jakob, 2000; Sakals and Sidle, 2004). This increase in landslide frequency and volume is largely related to the period of minimum rooting strength, caused by root strength deterioration after clearcut harvesting coupled with the slower recovery of root strength following replanting or natural regeneration (Example 2). Field

Box 2. Example 2: Landslides—timber harvesting

Root strength deterioration after timber harvesting may significantly increase landslide potential in steep forest sites. An example of changes in tree rooting strength for Sugi (*Cryptomeria japonica*) is presented in Fig. 2. The root decay curve for Sugi is based on uprooting tests on stumps conducted at different times after cutting (Kitamura and Namba, 1981). Root regrowth is also shown for regenerating trees based on the uprooting resistance power of different aged live trees (Kitamura and Namba, 1981). Root strength of harvested trees decays according to a negative exponential function (k and n are constants and t is years since cutting), whereas root regrowth of regenerating or invading trees follows a sigmoid recovery pattern (a, b, c, and k are empirical constants) (Sidle, 1991, 1992). Values of R and D (see equations in Fig. 2) are dimensionless and are scaled to 1 for maximum potential rooting strength. A maximum value of root cohesion for Sugi forests of 10 kPa was used based on data from Abe and Iwamoto (1987). Actual rates of root strength recovery are functions of species mix of regenerating vegetation, soil fertility, and other site conditions. Little information is available on root strength or root decay and regrowth of tropical and subtropical species. Net root strength is the sum of R and D; its minima are encountered about 3–15 yr after forest harvesting based on the time lag of root decay and the regeneration potential of the site (Sidle et al., 1985). It is apparent that Sugi loses about half of its initial root strength 5 yr after cutting. At about 12 yr after harvesting, replanted Sugi trees have regained about half of their potential rooting strength.

An example of how timber harvesting can affect soil water status and thus influence the occurrence of shallow landslides was simulated for tropical soils in Bukit Tarek, Malaysia, using synthesized annual rainfall based on daily records and seasonal distributions in the area (Sidle et al., 2004c). Assumptions were made for evapotranspiration, subsurface storm runoff, and deep drainage losses and rates (Bruijnzeel, 1990; Jones, 1997; Noguchi et al., 1997b) for shallow soils (1 m). It was assumed that a shallow landslide would initiate only when the soil was nearly saturated; this condition was not reached in the simulation. During the three largest storms of the year (75–100 mm), all preceded by wet conditions, soil moisture for forested conditions was only 9–13 mm lower than without the influence of trees. However, during a smaller storm (60 mm) preceded by dry conditions, the moisture content at the site with trees was 48 mm lower during the storm due to higher evapotranspiration. Thus, for wet antecedent periods it appears that logging would have only a very minor influence on landslide potential; for storms preceded by dry periods, timber harvesting may increase pore water pressure response (compared to unharvested forests). Another annual simulation of soil water accretion during storms for deeper soils (4 m), where an accumulation of 1650 mm of water was assumed necessary to initiate deep-seated landslides, showed that forest harvesting generated a total of 89 more days of deep mass movement compared to an uncut forest. As noted earlier, such differences in soil water levels after harvesting are likely short-lived in the tropics.

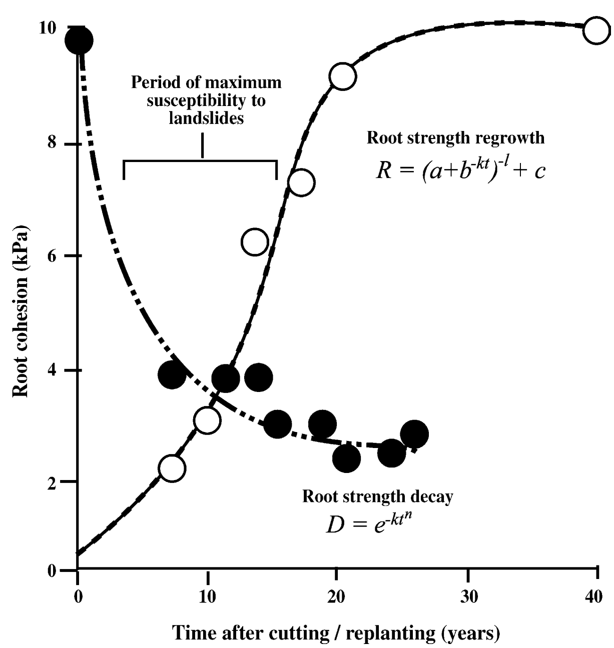

Fig. 2. Root strength decay and recovery curves for Sugi (Japanese cedar, *Cryptomeria japonica*)—based on uprooting tests of different ages of stumps and live trees, respectively. The root strength decay and recovery equations are from Sidle (1991, 1992).

observations of this phenomenon have been confirmed by independent tests on mechanical straining of roots (Burroughs and Thomas, 1977; Ziemer and Swanston, 1977; Wu et al., 1979; Abe and Ziemer, 1991). During rainstorms when hillslope soils are in a tenuous state of equilibrium, reinforcement from tree roots may provide the critical difference between stability and instability, especially when soils are partly or completely saturated (Sidle, 1992). The higher probability of landslides following timber removal diminishes substantially 15–25 yr after harvesting as replacement trees mature; thus, increased landslide rates should not persist throughout the entire rotation of the regenerating forest, except when stands are harvested at a very young age.

Modeling studies can assess the long-term effects of various silvicultural practices on landsliding. Simulations of landslide probability indicate that clearcutting forests, as well as combining clearcutting with later thinning operations, produce less stable conditions compared to partial cutting, selection cutting, and shelterwood harvesting systems (Sidle, 1992; Sakals and Sidle, 2004). Repeated harvesting cycles with progressively shorter rotations, which are being used in certain exotic plantations in the region (e.g., Ashton et al., 1998), suppress recovery of root strength (Dhakal and Sidle, 2003) and, thus, generate cumulative increases in the probability of landslide occurrence (Sidle, 1991, 1992). Additionally, logging methods that destroy understory vegetation or reduce the regeneration potential of new trees will increase landslide potential (Sidle and Wu, 1999). Longer intervals between initial and final shelterwood cuttings promote greater rooting strength than short intervals (Sidle, 1991).

Timber harvesting can also increase the water content of tropical soils by removing the depth-integrated transpiration via tree roots and by decreasing canopy interception losses. Forest cover can potentially buffer rainfall intensity (e.g., Keim and Skaugest, 2003) and concentrate some of the rainfall as stemflow, thus, focusing infiltration in areas of high root density (and strength) where preferential flow could mitigate pore water pressure accretion. Thus, if such soil moisture increases are substantial at steep sites after harvesting, a lower threshold of rainfall would be required to trigger shallow landslides. Additionally, the period and rate of seasonal activation of deep-seated landslide movement will be extended by increases in soil moisture (e.g., Iverson and Major, 1987). While no controlled studies have been conducted that compare soil moisture levels before and after timber harvesting in steep hillslopes of tropical Southeast Asia, post-logging responses in stream water yield (e.g., Bruijnzeel, 2004) and soil moisture (from other tropical areas; e.g., Parker, 1985) suggest that shallow increases in soil moisture are short lived (1–5 yr) due to rapid forest regeneration. Because of this rapid revegetation, the influence of any increases in soil moisture after logging on shallow landslide initiation may only be apparent for a very short period and the influence of forest cutting would likely be greatest during dry antecedent periods or during moderate storms in wet periods when the probability of shallow landsliding is low (Dhakal and Sidle, 2004; Sidle et al., 2004c).

4.2. Forest conversion to agriculture and exotic plantations

In contrast to timber harvesting with subsequent forest regeneration, forest conversion denotes a more or less permanent replacement of natural or secondary forest cover by another type of vegetation or land use—e.g., orchards, plantations, various annual agricultural crops, pasture, and brushland. Widespread conversion of tropical rainforests to oil palm, coffee, cocoa, fruit, tea, and rubber plantations has occurred or is occurring throughout much of Southeast Asia (Härdter et al., 1997; Craswell et al., 1998; Rosadi et al., 1999; Mattsson et al., 2000; D'haeze et al., 2005). Also, a variety of high cash value and subsistence crops, such as tomato, beans, pepper, taro, cabbage, rice, peanuts, cassava, and corn, are grown on steep hillslopes throughout the region. Where these plantations and crops are grown as monoculture, there is variable and generally less ground cover, reduced fertility, deteriorated soil structure, and insignificant rooting strength compared to tropical forests that previously occupied the hillslopes. Thus, the conversion of forests to agricultural lands can have significant effects on both surface and landslide erosion. In turn, accelerated erosion causes declines in site productivity, adversely affecting the sustainability of such conversion practices.

The rates of surface erosion on steep forest lands that have been converted to agriculture, single-story plantations, and pasture depend on the extent that dynamic management practices (with or without conservation measures) disturb and compact the soil, alter ground cover, and modify soil properties, thus lowering the rainfall-surface runoff threshold (Paningbatan et al., 1995; Chandler and Walter, 1998; van Noordwijk, 2000). The spatial extent of surface erosion in converted landscapes is also the product of the mosaic of land use activities and how these change in time and space related to the distribution and recovery of soil hydraulic properties and ground cover (van Noordwijk, 2000; Ziegler et al., 2004a). Landslide erosion in converted sites is more affected by the processes of root strength decay, undercutting or construction of steep slopes (e.g., terrace faces), and re-routing and concentration of water onto susceptible sites (Johnson et al., 1982; Sidle et al., 1985; Turkelboom, 1999).

Examples of soil loss estimates from various hillslope agriculture practices (mostly in converted forests) throughout Southeast Asia are presented in Table 3. The majority of these studies assessed only surface erosion at the field plot scale. Surface erosion rates from plots are generally higher than at the catchment scale because there is little opportunity for deposition and storage of sediment on uniform gradient plots. While few direct comparisons of scale can be made from data in Table 3, the catchment-scale sediment losses in Pulau Pinang and Cameron Highlands, Malaysia (Midmore et al., 1996; Wan Ruslan, 1997) and Java (van der Linden, 1983; Sinukaban et al., 1994, 1998; van Dijk, 2002) are generally (but not always) lower than erosion losses measured at hillslope, plot or bench-scales in similar areas (e.g., Hashim et al., 1995; Ongprasert, 1995; Poudel et al., 1999; van Dijk, 2002). However, for easily transportable soil particles, longer slope lengths may be necessary for rill and gully features to develop; thus, when such

Table 3
Examples of soil erosion for various soil management conditions and land covers in tropical Southeast Asia

Area	Methods/slope	Cover/treatment	Erosion ($t\ ha^{-1}\ yr^{-1}$)	Reference
Belalong Valley, Brunei	Plots: 30° slopes	Mixed dipterocarp forest with		Ross and Dykes (1996)
		1. Intact forest floor	1. 1.2	
		2. Root mat intact, litter removed	2. 23	
		3. Bare soil	3. 46	
Sunmberjaya, Lampung, Sumatra, Indonesia	Plots: 15° slopes	"Sun" coffee plantation with		Rosadi et al. (1999), Oki et al. (1999) (rates based on rainy season data that are annualized)
		1. No ground cover	1. 19–42	
		2. *Paspalum* ground cover	2. 0–1.5	
		3. Natural weed ground cover	3. 0.5–14	
Citere, West Java, Indonesia	Catchment	Vegetables	42–75	Sinukaban et al. (1994, 1998)
Upper Konto Region, East Java, Indonesia	Catchments	1. Mixed plantation forest	1. 0.4–4	Unpublished, cited by van Dijk (2002)
		2. Vegetables	2. 87	
		3. Mixed (agriculture and forest)	3. 10–12	
		4. Agriculture on bench terraces	4. 19–25	
Central Java, Indonesia	Catchment	Agriculture on bench terraces	12–14	van der Linden (1983)
Cikumutuk Catchment, West Java, Indonesia	Micro-catchment and catchment	Benched terraces with mixed crops, orchards, grasses and shrubs		van Dijk (2002)
		1. Bench scale	1. 24–202	
		2. Hillside scale	2. 11–70	
		3. Catchment scale	3. 47–73	
Cikumutuk Catchment, West Java, Indonesia	Terrace plots: 4.4–12°	Terrace beds with the following cover		van Dijk (2002)
		1. Bare soil	1. 71–118	
		2. Clean-weeded cassava	2. 30–123	
		3. Ginger	3. 30–34	
		4. Weeds	4. 11–24	
		5. Mixed crops	5. 1–31	
		Terrace riser (little or no cover)	59–393	
East Coast of Peninsular Malaysia, Kemaman (south Terengganu)	Large plots: 10° slopes	Agroforestry: cocoa and shade trees plus		Hashim et al. (1995)
		1. Intercropped with banana; bare ground	1. 70	
		2. Same as 1 with legume ground cover	2. 3.4	
		3. Monocropping; bare ground	3. 11.2	
		4. Monocropping; legumes	4. 1.0	
		5. Bare soil; no crops	5. 121	
Camaron Highlands Malaysia	Air photo assessment; landscape scale	Mixed vegetables with tea on prepared terraces (large area)	24	Midmore et al. (1996)
Pulau Pinang, Malaysia	Catchment various slopes	1. Orchards, agricultural crops + roads	1. 9.1	Wan Ruslan (1997)
		2. Urbanization/recreation	2. 31	
Malaysia	Various	Mature oil palm plantations	7.7–14	Maene et al. (1979), Morgan and Finney (1982), Lim (1990)
Baybay, Leyete, Philippines	Plots: 27° slopes	1. Bare	1. 69	Presbitero et al. (1995)
		2. Up/down slope cultivation: corn, sweet potato	2. 39	
		3. Same crops + contour cultivation with hedges (no intercropping)	3. 19	
		4. Same as 3 with rotation/intercropping of groundnut	4. 2.7	
Laguna, Philippines	Plots: 8–12° slopes	Corn and mungbean with		Paningbatan et al. (1995)
		1. Up/down slope cultivation	1. 140	
		2. Hedgerow alley cropping; contour till	2. 23	
		3. Same as 2 but with residue left as mulch	3. 2.8	
		4. Alley cropping + no till + mulching	4. 1.8	
Northern Mindanao, Philippines	Plots: 19–24° slopes	Cabbage, tomato and corn with		Poudel et al. (1999)
		1. Up/down slope cultivation	1. 65	
		2. Contour planting	2. 38	
		3. Strip cropping	3. 44	
		4. Contour hedge- rows of asparagus, pineapple, pigeon-pea, and lemongrass	4. 45	

Table 3 (*Continued*)

Area	Methods/slope	Cover/treatment	Erosion ($t\ ha^{-1}\ yr^{-1}$)	Reference
Southern Minanao, Philippines	Plots; steep slopes	Agroforestry: vegetable crops interplanted between double contoured rows of N-fixing shrubs and trees	3.4	Tacio (1993)
Mabini, Philippines	12–36 m plots: 8.5–11.3°	1. Corn and peanuts cultivated up and down the slope	1.44	Craswell et al. (1998)
		2. Same as 1 with hedgerows of banana and vetiver grass	2.5	
Khon Kaen, Thailand	Plots: 2.1–2.3° slopes	Rozelle (fiber crop) with		Sombatpanit et al. (1995)
		1. Up/down slope cultivation	1. 2.8	
		2. Hedgerow alley cropping; contour tillage	2. 3.0	
		3. Same as 2; residues mulched	3. 1.5	
		4. Same as 3 + no till	4. 0.3	
		5. Bare soil; no fiber crop	5. 48	
Mae Sa Mai, Chiang Mai, Thailand	30 m plot: 18–27°	1. Vegetables (two crop cycles)	1.167[a]	Ongprasert (1995)
	30 m plot: 10°	2. Vegetables (two crop cycles)	2. 25[a]	
Huai Luk, Chiang Mai, Thailand	36 m plot: 10–22°	Maize → beans (two cycles)	1[a]	Anecksamphant and Boonchee (1992); Anecksamphant et al. (1995)
Jabo, Mae Hong Son, Thailand	36 m plot: 17–22°	Maize → beans (two cycles)	50[a]	Inthapan et al. (1992)
Doi Tung, Chiang Rai district, Thailand	12–36 m plots: 11–29°	1. Coffee	1. 185 (40)	Craswell et al. (1998)
		2. Alley cropping (leucaena and pigeon pea)	2. 33 (21)	(numbers in parentheses
		3. 1 m wide Bahia grass strips	3. 28 (14)	are erosion losses for
		4. Hillside ditches	4. 19 (9)	fertilized crops)
		5. Upland rice cultivated up and down the slope (conventional farmer's practice)	5. 460 (193)	
Chiang Dao, Chiang Mai, Thailand	40 m plot: 15–20°	Maize, upland rice, beans, sesame	7	Unpublished, cited by Turkelboom (1999)
Pahka, Chiang Rai, Thailand	10–50 m plots: 17–36°	1. Upland rice	1. 60	Turkelboom (1999)
		2. Maize	2. 18	
		3. Beans	3. 10	
		4. Cabbage	4. 30	
Huai Thung Chao, Chiang Mai, Thailand	25 m plot: 28°	Upland rice	89[a]	Hurni and Nuntapong (1983)
Chiang Rai, Thailand (various sites)	30–40 m plots: 17–27°	Upland rice	61–90	Inthapan et al. (1992), Sombatpanit (1992), TA-HASDP (1990, 1991, 1992), unpublished, cited by Turkelboom (1999)
Upper Mae Tang River Basin, NW Thailand	Plots: 12° slopes	Rice paddy	3.9	Nishimura et al. (1997)
Tat Hamlett, Da River Watershed, northern Vietnam	10 m plots	1. Upland rice	1.78	Tran Duc Vien (1997)
		2. Cassava	2.41	
		3. Fallow	3.93	
Dak Gan, southern Vietnam	Unreported	Different ages of coffee plantations	102–107	Cited in D'haeze et al. (2005)
Throughout Southeast Asia	Catchment and plots	Undisturbed natural and secondary rainforest	0.01–1.2	Numerous sources within the region

[a] Median values determined by Turkelboom (1999).

concentrated erosion is a major source of sedimentation, longer slopes may have higher erosion rates. Additionally, catchment studies include effects of any roads and trails as well as landslide erosion; these compacted and disturbed surfaces tend to increase catchment erosion rates (e.g., Douglas et al., 1999). The complexities of scale effects on erosion processes together with the problems of different erosion measurement methods affect the values reported in Table 3; these problems have provoked much discussion in tropical agricultural environments (Stocking, 1995; Lal, 1998).

4.2.1. Surface erosion

Swidden (shifting) cultivation has been practiced for centuries in mountain lands of Southeast Asia as part of subsistence livelihoods (Spencer, 1966). Traditional swidden agriculture involves the clearing of forest patches in the dry season, subsequent burning prior to the rainy period to release nutrients, cultivation of the cleared patch for a period of years, fallowing, and secondary regrowth through various stages of succession (e.g., Spencer, 1966; Schmidt-Vogt, 1998; Fox et al., 2000). Original negative perceptions of shifting cultivation likely resulted from concerns about the growing replacement of primary tropical forests by secondary forests (e.g., Richards, 1964). Contemporary concerns are more likely related to land degradation arising from the gradual change from long-term fallow practices to semi-permanent cultivation systems with short-term fallow cycles; such an example has emerged in northern Thailand where uncleared lands are no longer abundant (Schmidt-Vogt, 1998).

Immediately following clearance, the residual effect of the forest on organic matter and aggregate stability keeps erosion rates relatively low (Morgan, 1995; Chappell et al., 1999). A rather steep (25°) swidden field in Sarawak, Malaysia, which was cleared of secondary forest, planted to rice, harvested, and then returned to fallow, had essentially the same rate of soil loss before (0.033 t ha^{-1} yr^{-1}) compared to after (0.038 t ha^{-1} yr^{-1}) clearance (Morgan, 1995). Other studies show that soil loss increases over time with repetitive tilling and planting on the same field. For example, Sinajin (1987) found an increase in soil loss from 0.87 to 2.02 t ha^{-1} between the first and third year of cultivation of cabbage on plots in Sabah, Malaysia. Similarly, soil losses for upland rice on new fields following clearing were about 1 t ha^{-1} yr^{-1}; after 12 yr of cultivation rates increased to 54 t ha^{-1} yr^{-1} (Kellman, 1969). Such increases in soil loss are likely related to a progressive breakdown in soil aggregates with repeated tillage (e.g., Turkelboom, 1999; Bronick and Lal, 2005).

Vegetable crops grown on moderate to steep slopes tend to have high rates of surface erosion (38–140 t ha^{-1} yr^{-1}) when cultivation is oriented up and down the hillslope, a typical practice in the region (Table 3). Cultivation along the hillslope contour together with agricultural hedgerows reduced erosion by half or more (19–45 t ha^{-1} yr^{-1}). Monoculture upland rice production is generally associated with high soil losses (60–90 t ha^{-1} yr^{-1}) when slope gradients exceed about 14°. One study conducted in northern Thailand found the highest erosion rates on fields where the crop cover was minimal during seasonal periods of highly erosive rainfall (see Example 3). A study of storm runoff response from zero-order agricultural basins in Leyte, Philippines, revealed that the largest percentage of overland flow (compared to subsurface flow) was generated on fallow pastures (previously planted to corn), followed by pasture-fallow with hedgerows and tilled corn with hedgerows, slash piling and mulching in cultivated cornfields, and recovering forest (previously slash and burn agriculture) (Chandler and Walter, 1998). The contributions of conservation practices (hedgerows and mulching), and especially reforestation, to hydrologic recovery support findings of lower surface erosion related to properly designed and implemented management practices and land covers.

While terracing has been widely used as a soil conservation practice in converted steeplands of Southeast Asia, it can be less effective in reducing surface erosion (rates 11–202 t ha^{-1} yr^{-1}, van der Linden, 1983; Midmore et al., 1996; van Dijk, 2002) than more passive measures, such as contour tillage, strip cropping, reduced tillage, and use of hedgerows. In at least one location in northern Thailand, high labor inputs for construction and maintenance of terraces were generally not compensated by additional crop yields, thus farmers opted not to maintain the practice (Turkelboom, 1999). Midmore et al. (1996) note that preparation and maintenance of terraces cut into natural hillslopes are major sources of the high erosion rates (estimated at 24 t ha^{-1} yr^{-1}) in tea plantations of the Cameron Highlands, Malaysia.

A detailed investigation of surface erosion sources in terraced agricultural land in West Java indicates that terrace risers (especially if poorly vegetated) are very significant sediment sources (59–393 t ha^{-1} yr^{-1}); on terrace beds soil loss ranged from 11 to 34 t ha^{-1} yr^{-1} for most cover crops, but increased to 80 and 94 t ha^{-1} yr^{-1} for clean-weeded cassava and bare soil terraces, respectively (van Dijk, 2002). The saturated hydraulic conductivity of surface soil on the terrace benches (260–426 mm h^{-1}, van Dijk, 2002) was within the range reported for many forest soils (Table 1), thus high rates of Hortonian overland flow and surface runoff would not be expected from the benches alone based on these data. Nevertheless, theoretically derived values of maximum infiltration capacity were much lower—ranging from 25–44 mm h^{-1} on compacted bare or weeded terraces to 72 to >233 mm h^{-1} on terraces with mixed crops (van Dijk and Bruijnzeel, 2004). These values are derived from a theoretical relationship between infiltration-excess rainfall and depth-averaged rainfall intensity assuming an exponential spatial distribution of maximum infiltration capacity (Yu et al., 1997; van Dijk, 2002). At the hillslope scale, soil losses declined due to deposition, and were similar in magnitude to outputs from the 125 ha catchment, which included other land uses (Table 3). Thus, erosion losses measured only on terrace benches will greatly underestimate sediment supply from these fields. The interaction of soil losses (including via mass wasting) from terrace risers with benches may be an important factor affecting the variability of sediment supply and erosion (Fig. 1c).

Contour strips are typically used by poor farmers to grow crops or trees for emergency subsistence; the alleys between the contour strips support more traditional agronomic crops. Effective erosion control depends on the extent of contact ground cover, the frequency and level of disturbance in the contour 'alleys', soil depth, and soil water storage capacity (e.g., Paningbatan et al., 1995; Presbitero et al., 1995). Low ground cover, frequent and deep soil disturbances, and shallow soils with low water holding capacities all promote surface erosion.

Oil palm plantations are probably the most rapidly expanding form of forest conversion in Malaysia and Indonesia during the past 40 yr (Härdter et al., 1997; Mattsson et al., 2000). While a few studies have documented moderate rates of surface erosion (7.7–14 t ha^{-1} yr^{-1}) from mature oil palm plantations (e.g., Maene et al., 1979; Lim, 1990), there appear to be no studies that clearly evaluate soil losses during the first few years following conversion. Installation of oil palm and other types of plantations, as well as certain vegetable crops, involve extensive land clearing, often with total removal of the topsoil (e.g., Wu, 1998; Lu et al., 2001) (Fig. 1c and d). Haron et al. (1998) note that organic matter in the topsoil of oil palm plantations is severely depleted following forest felling and site preparation, but recovers significantly 10 yr after plantation

Box 3. Example 3: Surface erosion—forest conversion

In Akha village (Pahka, Chiang Rai Province) in northern Thailand, rapid land-use change has accelerated both surface erosion and mass wasting (Turkelboom, 1999). Accelerated water-driven surface erosion was primarily in the form of rill and gully erosion. In contrast, a type of ravel erosion, whereby tilled, loose soil moves downslope via gravity in dry conditions, was the dominant process in steep (>35°) fields of relatively short length. Erosion at all spatio-temporal scales was influenced by household strategies that: (1) determine cropping activities (timing and type); (2) generate surface runoff on disturbed surfaces; and (3) dictate the fate of surface runoff, once initiated.

Hillslope fields were typically subjected to a critical period of high erosion risk extending from the time of clearing for planting to the time when adequate cover develops to protect the soil surface from erosive rainfall. The threshold for total contact cover (including weeds, rocks, debris) was about 50%. Of the four crops studied, upland rice had the longest *at-risk* period: ≈120 days (Fig. 3a). The *at-risk* period for maize, soy beans, and cabbage was about 40 days. Six of seven observed fields with erosion rates >100 t ha^{-1} $cycle^{-1}$ were planted with upland rice; 11 of 12 monitored bean or maize fields did not have substantial rill networks. High erosion rates on upland rice fields are facilitated by late sowing and slow cover development relative to the onset of erosive storms associated with the wet monsoon season (May–October).

Generally, slopes >30° and longer than 25 m were subject to rill erosion. If rills and gullies did not develop, soil losses were usually <2 t ha^{-1} $cycle^{-1}$. Raindrop-induced surface sealing, construction of in-field furrows, and compaction by field maintenance all facilitated infiltration-excess overland flow, which in turn contributed to rill formation and eventually rill incision. Compaction occurring toward the end of the growing period (via raindrop impact, during weeding) also made fields susceptible to infiltration-excess overland flow. Gullies were persistent sediment sources, runoff concentration agents, and pathways for run-on water entering fields. The highest erosion rates occurred when gully and plow-layer erosion occurred together (soil loss on four fields: 180–350 t ha^{-1} $cycle^{-1}$). Plow-layer erosion coincided with the development of saturation overland flow above a plow pan, which decreased shear resistance to the point where topsoil moved downslope by a combination of concentrated surface water and gravity erosion. Fields with only gullies and/or rills typically had soil loss rates ≤64 t ha^{-1} $cycle^{-1}$ (rates from four fields).

In a general survey, 28 of 59 fields had gullies created from overland flow generated on disturbed areas. Agricultural surfaces, dwelling sites, paths/roads, and landslide scars contributed 37, 27, 16, and 9% of the basin-wide infiltration-excess overland flow volume (Fig. 3b). Roads and paths, which occupy less than 1% of the basin area, generated a disproportional volume of this overland flow. Paths alone were by far the most important source of run-on water entering fields. Other sources of field run-on included irrigation channels, paddy fields, water diversions, and landslide planes—features affected by surface water management.

Although field-generated runoff was comparatively high, overland flow from one field to another was rare owing to the isolation of fields and the typical presence of downslope fallow vegetation. Despite some degree of buffering provided by fallow vegetation, much surface runoff and entrained sediment from all dry-land areas reached the stream system via a complex network of rills, gullies, paths, and diversions. For example, runoff generated on paths and about 50% of the settlement area drained directly into the stream. The types of surface erosion processes and sediment delivery mechanisms occurring at Pahka are representative of those in many upland communities in montane Southeast Asia.

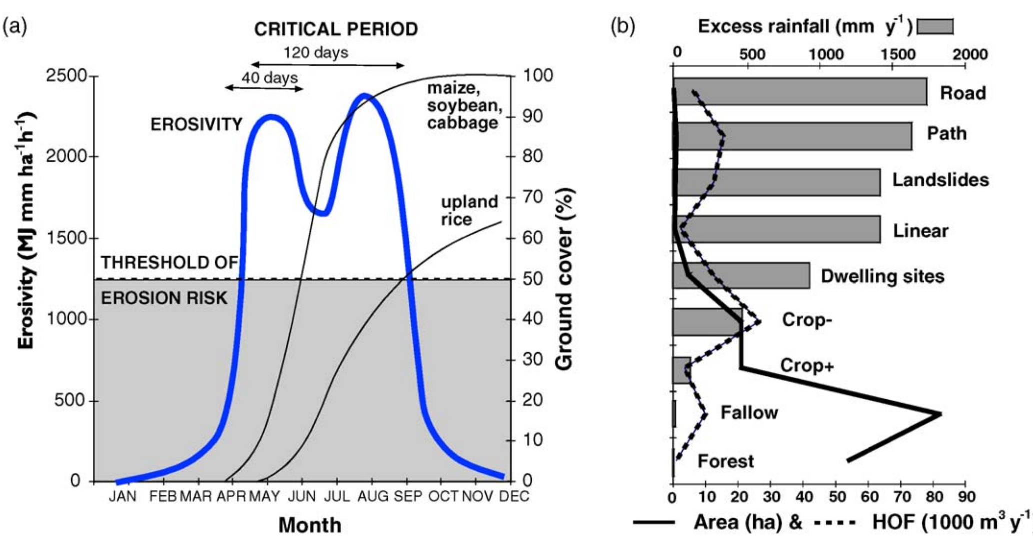

Fig. 3. (a) Comparison of timing of erosive rainfall (thick line) and cover development for upland rice vs. other crops (thin lines); the critical period for in-field erosion risk in Pahka, which occurs before the crop cover reaches 50%, is 120 and 40 days for upland rice and other crops, respectively. (b) For each surface condition the following data are shown: area within the basin (solid line); contribution to total basin-wide Hortonian overland flow (HOF, dashed line); and excess rainfall generated (shaded bar). Data are for 1995 in the 168 ha Dze Donglo catchment at Pahka. Total rainfall and total HOF are 1833 mm and 137,000 m^3, respectively. Linear features include irrigation channels, drainage ditches, and diversions; the "+" and "−" signs indicate whether the crop is above or below the 50% threshold for erosion risk.

establishment. During the period of disturbance associated with land clearing, much of the natural soil structure is destroyed; soil particles are easily detached from such disturbed sites and areas compacted by machinery facilitate overland flow. In areas where fire is used in the forest clearing process, additional runoff and erosion (including dry ravel) typically occurs during the first few years after burning (Heede et al., 1988; Scott and Van Wyk, 1990; Bruijnzeel, 2004; Sidle et al., 2004a).

In general, widespread vegetation clearing and subsequent burning on steep hillslopes facilitate both surface and landslide erosion processes. Fire can cause hydrophobicity (e.g., DeBano, 2000), thus promoting overland flow, surface erosion, and dry ravel. As is true for any form of cultivation, intensive agronomic practices decrease the infiltration capacity of the soil, increasing the propensity for surface runoff and surface erosion during storms (Ataroff and Monasterio, 1997; Roose and Ndayizigiye, 1997; Ziegler et al., 2004a). Sparse crop cover that leaves a large portion of the field unprotected during highly erosive rainfall periods promotes splash and rill erosion (Turkelboom, 1999). The patchiness of many small-scale, traditional shifting cultivation systems (Schmidt-Vogt, 1998; Fox et al., 1995; Turkelboom, 1999) may allow overland flow that is generated from these fields to reinfiltrate, and surface eroded sediment to redeposit, in less disturbed areas downslope.

4.2.2. Landslides

Compared to surface erosion, the effects of forest conversion on landslides in Southeast Asia have been much less studied. Progressive forest clearing/conversion and deterioration of the land base in developing countries of Africa, Asia, and Latin America has been associated with general increases in landslides (Haldemann, 1956; Harwood, 1996; Fischer and Vasseur, 2000), although specific rates of increase have not been reported. Lanly (1969) estimated that the land base in the rain forest of the Ivory Coast declined by 30% from 1956 to 1966 because of shifting cultivation; these losses were significantly related to landslides. While few studies in Southeast Asia have assessed the extent of landsliding on cultivated hillslopes, scattered reports of such occurrences appear in reports of other types of degradation or land management systems and in GIS-based spatial hazard analyses in the tropics (Gupta and Joshi, 1990; Turkelboom, 1999; Perotto-Baldiviezo et al., 2004; Sidle et al., 2004c). Some of surficial erosion features (e.g., tillage erosion and plow layer erosion) identified by Turkelboom (1999) on steep cultivated slopes of northern Thailand, are actually mass erosion processes (i.e., gravity-induced ravel and small-scale flow-type failures, respectively).

Because of the lack of long-term, comprehensive studies on landslide erosion in Southeast Asia, uncertainties exist related to effects of land cover change. For example, several reports of the November 1988 landslides during an intense monsoon storm in the southern peninsula of Thailand yielded somewhat different conclusions related to the role of converted forest cover on landslide erosion. DeGraff (1990) suggested that forest conversion to weaker-rooted rubber plantations was responsible for a higher level of landsliding than would be expected with native forest cover. In the same general area, Tang (1991) concluded that the catastrophic landslides and debris flows resulted from a combination of extreme rainfall, destruction of natural forest cover, and conversion to rubber plantations in thin, granitic soils. Conversely, Phien-Wej et al. (1993) noted that landslide density appeared independent of vegetation cover, implying that the storm magnitude overwhelmed the stabilizing influence of the different root strengths. Thus, the extent to which forest conversion to rubber plantations in the region contributed to the landslide catastrophes was unclear because long-term inventories were not available; this conversion may have enhanced the rapid and frequent mobilization of landslides into debris flows, which contributed to the high loss of life and destruction (Harper, 1993).

The major effect of conversion of forest land to agriculture on landslide erosion can be attributed to a more or less permanent loss in rooting strength compared to previous forest or brush vegetation (Sidle et al., 1985; Sidle, 2005). Shallow-rooted agronomic species and grasses have negligible rooting strength compared to deeper-rooted trees and shrubs (Rice et al., 1969; Marden and Rowan, 1993; Bergin et al., 1995). Thus, long-term reductions in root strength following forest conversion can be translated into an increase in landslide probability (Sidle, 1992). However, the duration and/or spatial extent of most erosion investigations are typically not large enough to capture landslide erosion; thus, it is not included in most erosion estimates unless indirectly accounted for in catchment outputs. Even for catchment studies, unless bedload sediment is measured, much of the landslide erosion will be missed because landslides contribute proportionally more bedload than suspended load compared to surface erosion, and significant bedload transport only occurs episodically during the largest storms of each year (Sidle, 1988; Anthony and Julian, 1999; Gomi and Sidle, 2003). Evidence of landslides related to conversion of steep forested hillslopes to coffee plantations (Sumatra, Indonesia) and shallow-rooted secondary vegetation (northern Thailand) are shown in Example 4 (also see Fig. 1a).

In addition to the weaker rooting strength of agricultural crops or plantations following forest conversion, terraces constructed on steep slopes can increase landslide potential by both concentrating water on the terraces (especially if back-sloped) and by oversteepening the terrace face (e.g., Johnson et al., 1982; Turkelboom, 1999). Terraces that impound water are especially susceptible to landslide erosion in steep terrain (see Example 4). Evidence of landslides on terraces has been reported in several regions of Asia (Johnson et al., 1982; Billard et al., 1993; Turkelboom, 1999; Sidle and Chigira, 2004), but, in general, the practice is viewed positively from a soil conservation perspective by most management agencies and donors. At the plot scale, such a perspective may be reasonable for surface erosion, however, at the catchment scale, especially when large terraces are constructed (including the requisite road systems), some practices may actually cause a net sediment gain by creating potential sites of instability (e.g., Johnson et al., 1982) (Fig. 1c).

Box 4. Example 4: Landslides—forest conversion

Within a 2.5 ha catchment in Sumber Jaya (Sumatra, Indonesia) that was converted from tropical forest to coffee plantation in 1986, apparent increases in landslides were observed. Three shallow landslides (one a combination landslide-debris flow) occurred 14–16 yr after forest conversion (Table 4). All three landslides occurred on the side of the catchment that was converted in 1986; the other side of the catchment, cleared in 1998, has not yet experienced slope failures. While other factors such as road drainage and slope undercutting likely triggered some of these landslides, the reduction in rooting strength predisposed the hillslopes to failure. Although none of these landslides were particularly large (31–113 m^3), the annualized average soil loss (assuming a soil bulk density of 1.1 g m^{-3}) from landslides during the 4-yr monitoring period (2000–2003 inclusive) was 27 t ha^{-1} yr^{-1} (Table 4). This value is in the same range as measured rates of surface erosion from coffee plantations with no ground cover. Such landslide erosion has typically not been considered in soil loss estimates from plantations and obviously represents a substantial source of sediment. Much of this landslide sediment reached the headwater stream of this small catchment either immediately after failure or within a few years. While there was no long-term control catchment to compare these landslide losses against, no landslides were observed in nearby forests of similar gradient during the period from 2000 to 2003. Thus, all landslides appear to be related, at least in part, to the reduced rooting strength of the coffee plantation compared to the original tropical forest cover (see Fig. 2).

In the Dze Donglo catchment (Pahka, Chiang Rai Province) in northern Thailand, 16 landslides occurred in 1994 in response to three relatively large, long-duration rainfall events (54–94 mm) (Table 5). Although not unusual (all storms had return periods <1 yr), the events occurred following a period of heavy rainfall in the second half of the monsoon wet period when antecedent soil moisture was high. The 6310 m^3 of material mobilized by these events equates to 41 t ha^{-1} yr^{-1} of landslide erosion in the 168 ha basin, slightly higher but of similar magnitude compared to the rates reported in the coffee plantation in Sumber Jaya. Most of the landslides were shallow translational failures occurring on steep concave hillslopes near stream channels. While the primary failure mechanism was undercutting of the toe by the stream, the role of forest conversion cannot be ignored. Secondary vegetation, including bamboo, was dominant on the failed hillslopes. In comparison with the original forest tree species, shallow rooted secondary vegetation has limited ability to anchor into stable substrate, and it evaporates less water from the deep subsoil, thereby rendering the weaker, permeable soil more susceptible to pore water accretion during high intensity or long duration storms. The other six failures were also likely influenced by vegetation conversion and related activities. The two other translation slides (total volume 420 m^3) not only had shallow-rooted secondary vegetation, but also incised footpaths that likely reduced the toe support of the hillslope. Sites of the less common slump-earthflow type failures were characteristically deeper and more gently sloping than the translational landslides. Infiltrating water from paddy fields and an irrigation channel contributed to the development of high pore water pressures prior during the period of heavy rainfall that initiated these deeper failures. Additionally, the paddy fields were located immediately above steep river banks, where substantial seepage contributed to the undercutting of the supporting cliff face. Aside from the general on-site degradation caused by these landslides (area affected was 0.9 ha), material mobilized by 13 of the landslides deposited directly into the stream channel. Additionally, the landslide scars became the source for about 9% of the erosion-producing Hortonian overland flow throughout the basin over the 2-yr study period. Thus, the sediment mobilized by all 16 landslides represented a persistent sediment source.

Table 4
Landslides type, volume, and probable cause, Sumber Jaya, Indonesia (Sidle, unpublished)

Landslide (year)	Volume (m^3)	Gradient (°)	Failure type	Probable cause
2000	113.1	39	Debris slide/debris flow	Road drainage and conversion of forest to coffee plantation
2001	101.3	36	Debris slide/debris flow	Road drainage and conversion of forest to coffee plantation
2002	31.2	47	Debris avalanche	Undercutting of bank and conversion of forest to coffee plantation

Table 5
Landslides type, volume, and probable cause, Dze Donglo catchment, northern Thailand

Number of landslides	Volume (m^3)	Gradient (°)	Failure type	Probable cause and contributing factors
10	610	40–50	Translational slide	River incision and/or subsurface storm flow during the wet period
2	420	35–41	Translational slide	Fallow vegetation; concave; steep slopes; runoff from footpath
3	5260	15–24	Slump-earthflow	Soil saturation of wetland/paddy
1	25	6	Small earthflow	Sautration from old irrigation channel

Parent material is granite and phyllite.

4.3. Agroforestry

4.3.1. Surface erosion

The modern practice of agroforestry includes the simultaneous cultivation of annual crops and trees, with the possible inclusion of animal grazing. While at the land tenant scale, such systems offer both socio-economic and biophysical benefits related to sustainability, there is little information on cumulative effects of these practices at larger scales (Steiner, 1988; Fischer and Vasseur, 2000). The forestry component of agroforesty systems offers benefits for surface erosion control due to maintenance of organic matter, soil structure, infiltration, and microbial biomass (Rai and Sharma, 1998; Islam and Weil, 2000; Ong et al., 2000). The agricultural component of these integrated systems may provide soil cover, but sometimes at the expense of long-term physical and chemical properties of the soil depending on the amount of cultivation required.

The incorporation of a cover crop within a plantation of trees, or between rows of trees or shrubs, has been demonstrated to be an effective control on surface erosion in Southeast Asia (Tacio, 1993; Hashim et al., 1995; Oki et al., 1999; Rosadi et al., 1999). Erosion rates for coffee (19–185 t ha^{-1} yr^{-1}) and cocao (11–121 t ha^{-1} yr^{-1}) trees with no ground cover were very high, but rates are reduced to 0–3.4 t ha^{-1} yr^{-1} when cover crops were incorporated (Hashim et al., 1995; Craswell et al., 1998; Rosadi et al., 1999; D'haeze et al., 2005) (Table 3). Without ground cover, the generally large raindrop size from tree canopy drip can cause soil detachment (Nanko et al., 2004). The upper end of these reduced erosion rates from improved agroforestry practices are in the same range as average mass erosion from clear-cut forests in steep terrain (Sidle et al., 1985). The inclusion of nitrogen-fixing ground cover, trees, or shrubs is beneficial for long-term site maintenance of agroforestry systems (Tacio, 1993; Hashim et al., 1995). To prevent high rates of surface erosion, agroforestry systems require careful selection of both crops and tree species (Morgan, 1995). Additionally, as with other hillslope agricultural practices, erosion increases when inappropriate conservation measures are implemented. A relatively sustainable system of cinnamon trees with a legume ground cover in Sumatra is shown in Fig. 1e.

4.3.2. Landslides

The effects of agroforestry practices on landslide erosion are much the same as for forest conversion to agriculture—i.e., decreased rooting strength and the occasional incorporation of terraces (that are subject to failure) in steep terrain increase the probability of landslides (Sidle, 2005). However, depending on the density of tree plantings in agroforesty systems, rooting strength of the site may be higher than in strictly cropland or pasture conditions, albeit less then typical rooting strength of mature forests (Fig. 4). The off-site consequences and cumulative effects of agroforestry practices at the catchment scale on both surface and landslide erosion have not been studied nor have any systematic investigations evaluated the effects of this management system on landslide initiation.

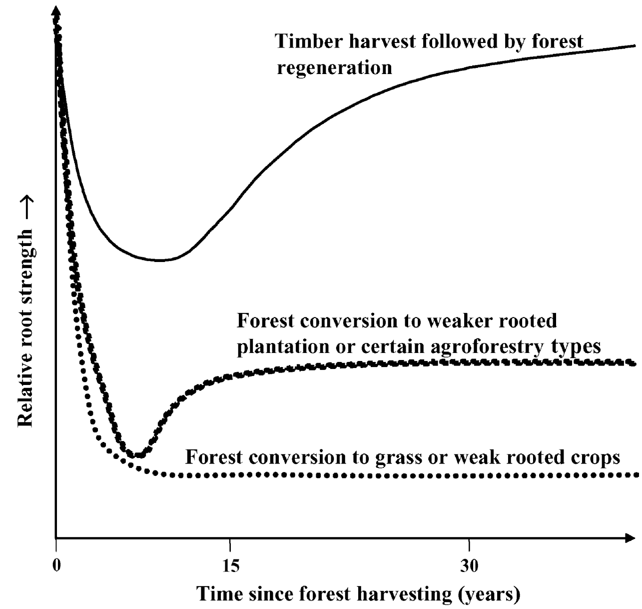

Fig. 4. Hypothetical changes in vegetation rooting strength related to scenarios of timber harvesting and subsequent forest regeneration, forest removal and conversion to plantations or agroforestry, and conversion of forest to grassland (modified from Sidle and Dhakal (2002)).

4.4. Grazing

The effects of domestic animal grazing and other animal trampling on surface erosion and localized slope failures in Southeast Asia are apparent in many areas, but the aggregate contributions of these impacts have not been assessed. One reason for this poor documentation is that animal traffic and grazing commonly occur together with other agricultural and rural activities (e.g., transport, cultivation, household activities) (Islam et al., 2001; Rasul and Thapa, 2003). As such, soil compaction, disturbance, and crusting as well as vegetation removal and species composition change caused by domestic animals is intrinsically linked with other environmental impacts.

4.4.1. Surface erosion

While some investigations contend that the effect of grazing on surface erosion is small (e.g., Rowntree et al., 2004), numerous process-based investigations have documented decreases in vegetative cover and infiltration capacity, as well as increases in soil crusting, bulk density, and sediment levels in runoff related to grazing; these findings clearly show accelerated erosion associated with grazing intensity (e.g., Gifford and Hawkins, 1978; Warren et al., 1986; Greene et al., 1994; Roth, 2004). In one of the few such studies conducted in Southeast Asia, Chandler and Walter (1998) found that grazed pastures in the Philippines had the highest amount of surface runoff compared to tilled, mulched and forested covers. Thus, in cases where large tracks of forest land are cleared for grazing, increases in surface erosion can be expected (e.g., Lebaron et al., 1979; Karambiri et al., 2003) and will be proportional to the intensity of grazing, especially when this occurs during the wet season. Even in a relatively flat (≈0.6° longitudinal gradient; 1.4 ha) catchment in the Sahel, grazing pressures elevated sediment yields to 4.0–8.4 t ha^{-1} yr^{-1} (Karambiri et al., 2003).

4.4.2. Landslides

Converting forest or brush land to pasture in steep terrain has been shown to increase landslide erosion in many parts of the world (Haldemann, 1956; Fairbairn, 1967; Rice et al., 1969; Temple and Rapp, 1972; Luckman et al., 1999) due to the negligible deep rooting strength of grasses compared to that of trees (Fig. 4). Animals congregating around streams and water bodies represent an important cause of erosion and sediment transport by disturbing and compacting soils, and removing vegetation in wet riparian corridors, as well as destabilizing streambanks by their overburden effects (e.g., Sidle and Sharma, 1996). Runoff and erosion produced from upland compacted pastures (e.g., Chandler and Walter, 1998) can contribute to initiation of debris flows in wet areas downslope (e.g., rice paddy fields). Small bank failures along overgrazed stream channels are common in many parts of Southeast Asia (Fig. 1f). In addition to the more widespread trampling effects on hillslopes, animal trails provide similar (if not greater) sources of erosion and sediment compared to footpaths (see next section); animal trails are highly compacted and typically directly linked to streams.

4.5. Roads and trails

In steep terrain, roads and trails contribute the most sediment loss per unit area within the catchment (Sidle et al., 1985; Malmer, 1996b; Froehlich and Walling, 1997; Larsen and Parks, 1997; Ziegler et al., 2000; Wemple et al., 2001; Sidle et al., 2004b). The modification of hydrologic pathways by roads and trails affects both surface erosion and landslide processes, but in different ways. Thus, the hydrogeomorphic characteristics of the site must be clearly understood to assess effects of roads and trails on these erosion processes and sediment pathways.

4.5.1. Surface erosion

Roads and trails modify site hydrology by decreasing the hydraulic conductivity and infiltration capacity of the traveling surface (Table 1), redirecting incoming rainfall and water as Hortonian overland flow, and concentrating this runoff into various parts of the catchment (e.g., Megahan, 1972; Baharuddin et al., 1996; Ziegler and Giambelluca, 1997; Ziegler et al., 2004b). Additionally, roads cut into hillslopes may intercept subsurface water and reroute it along the road surface or through surface drains (e.g., Megahan, 1972; Wemple et al., 1996). The extent of subsurface flow interception depends on the existence and depth of a hydrologic impeding layer below the hillslope soil. If the restricting layer is exposed at the road cut then a greater proportion of the subsurface flow will be intercepted by the road in contrast to where the restricting layer is below the road cut. The design and location of roads and paths determines whether surface water generated by these various mechanisms will exacerbate surface erosion, landslides, and peak flows (e.g., Packer, 1967; Sidle et al., 1985, 2004b; Megahan, 1987; Douglas et al., 1999). Drainage systems on developed roads can alleviate certain adverse consequences; however, drainage outlets can be sites of gully and landslide initiation if poorly located (e.g., Sidle et al., 1985; Piehl et al., 1988; Wemple et al., 1996).

In Southeast Asia, most mountain roads and trails are not built according to any design engineering standard, are not surfaced with rock, and drainage systems often do not exist; thus, surface runoff simply follows the contour of the road and discharges onto slopes or into channels at topographic low points or discharge nodes (Fig. 5). Where soil or substrate of the roadbed is erodible, the lack of drainage systems can have catastrophic consequences for sediment production, particularly in steep terrain. Baharuddin et al. (1995) measured surface erosion losses from logging roads and skid trails in a tropical rainforest in Malaysia of 13.3 and 10.1 t ha^{-1} yr^{-1}, respectively, during the first year after logging. In the second year, these respective losses declined to 3.1 and 2.1 t ha^{-1} yr^{-1}. Much higher soil losses were observed on roads accessing agricultural sites in northern Thailand (120 t ha^{-1} yr^{-1}, Ziegler et al., 2004b), skid trails in Sabah (30–104 t ha^{-1} yr^{-1}, Hartanto et al., 2003; 77–547 t ha^{-1} yr^{-1}, Malmer, 1996b), and logging roads and skid trails in Peninsular Malaysia (272 and 275 t ha^{-1} yr^{-1}, respectively, Sidle et al., 2004b). Malmer (1996b) noted a three- to five-fold decline in annual sediment production from skid trails between the first 2 yr after logging compared to the third year. An example of storm runoff and erosion fluxes generated from a forest road in Peninsular Malaysia is given in Example 5.

Paths in managed forests and especially in converted agricultural lands typically produce less sediment than roads, but they constitute important sediment sources (e.g., Ziegler et al., 2000). In one site in northern Thailand, paths and dwelling sites produced roughly 2–3% of the total catchment

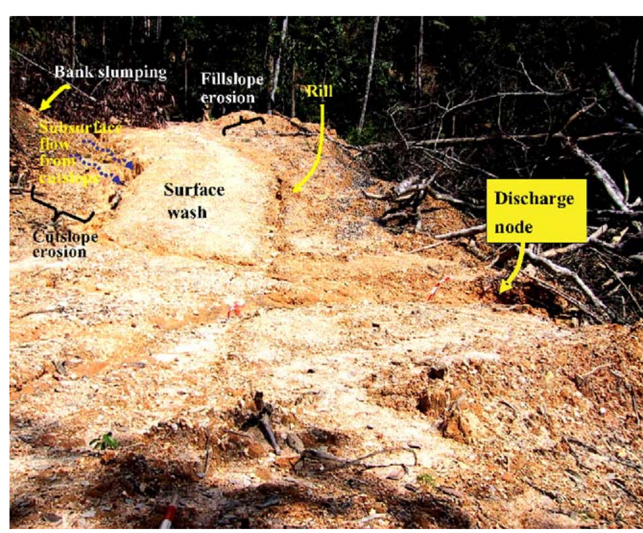

Fig. 5. Rill and gully erosion on a logging road in central Malaysia. Erosion features indicated on the photograph converge at a discharge 'node' located at a topographic low point on the road. Water and sediment discharged at this node connect directly to a headwater stream.

Box 5. Example 5: Surface erosion—roads

Within Catchment 3 (C3, 13 ha) at the Bukit Tarek Experiment site in Peninsular Malaysia, a 51.5 m mid-slope road segment was instrumented with two v-notch weirs (at discharge nodes); one v-notch weir was installed at the outlet of a zero-order basin (hollow) that was truncated by the road. Sediment samples were manually collected during storms to estimate surface erosion. The road cut is >2 m deep in places and typically extends into the bedrock below the soil. A series of four storms was monitored during a 4-day period (2–5 December 2002). Prior to the first storm (2 December), the site was quite dry for this season.

During the first three storms the catchment began to 'wet up' and almost no cutslope interception or outflow from the hollow was produced; essentially all runoff could be attributed to Hortonian overland flow (infiltration-excess overland flow). However, by the December 5th storm, enough water accumulated in the upslope soils (including the hollow) to generate extensive amounts of subsurface flow and discharge from the hollow (both intercepted at the road cut; Fig. 6a). The initial peak in the composite runoff hydrograph (Fig. 6b) is generated by Hortonian overland flow, while the much longer secondary peak (and slower recession) is generated by interception of subsurface flow and discharge from the hollow. These intercepted flows can increase peak discharge in the stream as well as deplete downslope soil moisture recharge. The maximum measured sediment flux from this road segment was 87 kg h^{-1}, and the total amount of sediment transported during the rising and peak portions of the hydrograph was 58.9 kg (Fig. 6b). An estimated additional 4.7 kg was transported after the last sediment sample was collected, making the total sediment production for the storm about 63.6 kg.

If we assume that the entire 0.69 km road system in Bukit Tarek C3 delivered similar levels of soil loss, about 0.85 t of solids were transported from the road system alone during this intense event, representing a total solids yield from the road of >2 t ha^{-1}. This storm occurred about 3 yr after initial road construction; the solids yield from this single December 5th event is about 0.5% of the total estimated soil loss during the 16-month period immediately after road construction (when erosion should have been much higher) (Sidle et al., 2004b). The off-site transport of sediment is strongly influenced by discharge nodes along the road that directly connect to headwater channel segments (Fig. 5).

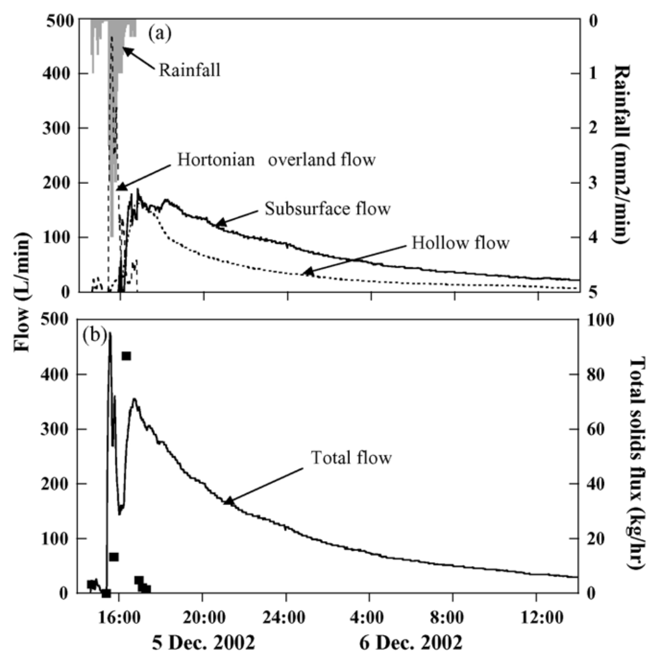

Fig. 6. (a) Separation of road runoff into Hortonian overland flow, intercepted subsurface flow, and flow intercepted from a truncated geomorphic hollow, during a major storm event on 5 December 2002, Bukit Tarek Catchment C3, Malaysia. (b) Composite road runoff hydrograph for the 5 December storm together with measured fluxes of total solids.

sediment (Ziegler et al., 2004a). Surface erosion from trails in East Java was approximately 70 t ha^{-1} yr^{-1} (Rijsdijk and Bruijnzeel, 1990). If paths are located within forests, their influence on sedimentation may be small because runoff will be buffered by downslope vegetation; however, if located in converted plantations and agricultural fields, paths may capture sediment-laden runoff from adjacent fields and redirect it to streams, or divert runoff onto fields where it moves to streams as overland flow (Ziegler et al., 2004a). Two studies in northern Thailand showed the importance of paths and in-field furrows as the sources of overland flow and surface erosion. Actual sediment delivery to the stream, however, was dependent on the position of the field relative to the stream network, downslope vegetative buffers, and/or features (e.g., gullies, diversions) that allowed surface flow to bypass potential buffers (Turkelboom, 1999; Ziegler et al., 2001). At the Pahka site, paths generating only about 11% of the annual estimated basin-wide Hortonian flow were the single most important cause of runon water in agricultural fields, generating substantial rill and gully erosion (Turkelboom, 1999). While little information is available on the cumulative impact of footpaths in Southeast Asia, they are obviously important routes for farmers and local hill tribe people (Ziegler et al., 2001, 2004b) (Fig. 1g). The two Thailand cases indicate that even if all compacted surfaces (roads, paths, dwelling sites) within a catchment occupy a small fraction of the total area, they can still contribute disproportionately to overland flow generation. Example 6 shows the dynamic runoff and sediment fluxes produced from a small footpath during a high-intensity monsoon in northern Thailand.

The findings cited here, as well as evidence presented in Example 6, indicate that in addition to path density, the adjacent topography and land use that affects runon during major events must be considered. Other factors that may enhance erosion from footpaths and sediment delivery to streams include: (1) paths oriented perpendicular to hillslope contours; (2) interconnectivity of path networks (including with roads); (3) direct

Box 6. Example 6: Surface erosion—footpaths

Erosion losses and runoff from two agricultural footpath segments (each about 7 m in length with slope gradients of 11° and 23°) in Amphoe Mae Taeng catchment, northern Thailand are presented for a 1 h monsoon storm on 25 August 2002 (Fig. 7). The maximum concentration of total solids (mineral + organics) in path runoff was 136 g L^{-1}; total storm rainfall was 30.5 mm and maximum 5 min intensity was 96 mm h^{-1}. While the upper, steeper path segment had about twice the sediment concentration throughout most of the storm, the lower (gentler) path generated more runoff, especially during the peak of the runoff hydrograph (Fig. 7). The higher total solids concentrations from the upper path are likely related to the steeper gradient, while the higher peak runoff from the lower path is due to significant runoff contributions from the neighboring cornfield which were shunted onto this section during the event. At the peak of the storm the contributing area of the lower path expanded more than 4-fold (compared to the original path area) in contrast to only a 2.4-fold expansion in the upper path segment.

Based on the hydrograph and samples collected during this intense event, 14.3 kg of solids were transported from only 14 m of footpaths (7.8 m^2 of total path area) (Fig. 7). Thus, expressed as solids yield on the basis of the path area alone, this represents 18.3 t ha^{-1}. Based on the maximum contributing area (i.e., including adjacent cornfield contributions), this surface erosion is still 5.5 t ha^{-1}, higher than the yield calculated for 5 December 2002 storm from the logging road in Malaysia (expressed on a unit contributing area basis). In either case, these are very high values for just one storm, and it is apparent that the path facilitates the transport of locally eroded material from the adjacent cornfield.

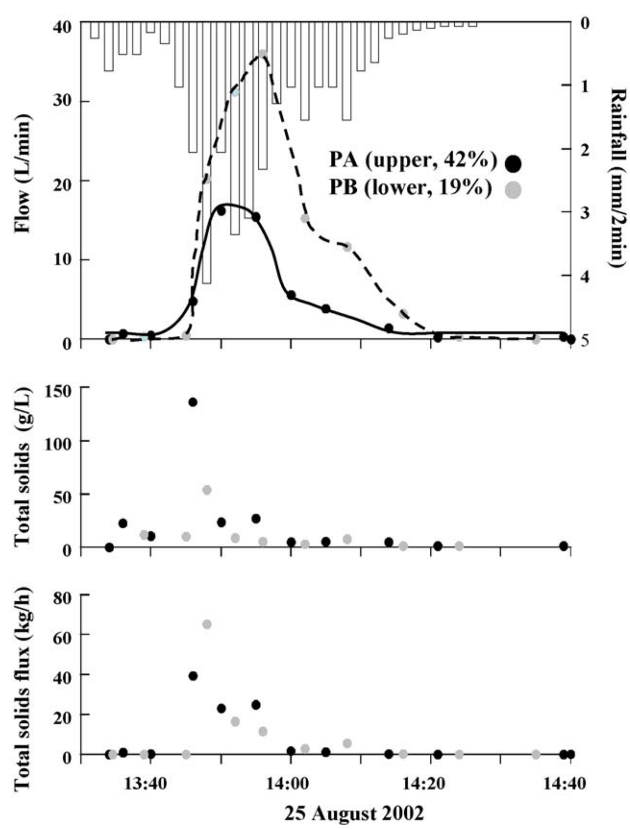

Fig. 7. Runoff from two footpath segments with average gradients of 23° (●) and 11° (◎), respectively (a), along with respective total solids concentrations (b), and fluxes of solids (c), during an intense, 1 h monsoon storm in a converted agricultural catchment in northern Thailand.

Box 7. Example 7: Landslides—forest roads/skid trails

In late 2003, a little used forest road was activated and new trails were established for the purpose of clearing the remaining forest stand and establishing a forest plantation in Bukit Tarek Catchment C3, Peninsular Malaysia. Six months after logging, 16 landslides were surveyed along newly constructed skid trails and the widened logging road, generating a total of 93.7 m^3 (95.6 t) of sediment (Fig. 8). The causes of the landslides were slope undercutting (13 cases) and overloading of fill material (3 cases). While much of this sediment remained on the skid trials and forest road, the annualized soil loss (14.7 t ha^{-1} yr^{-1}) from these recent landslides is actually higher than the estimated surface erosion from the original logging road, 10.8 t ha^{-1} yr^{-1} (measured over a 16-month period). These values are both calculated based on the total catchment area (13 ha) and would be much higher if only the affected road or skid trail area was considered.

The high density of skid trials created a terraced effect on the hillslope (Fig. 8), however these 'terraces' were not effective in reducing erosion (including the mobilization of landslide sediment) and runoff. On the contrary, as witnessed in the field during subsequent monsoon storms, these trails provided direct conduits for water and sediment that connected to the road and eventually the stream. Thus, much of the landslide sediment will be mobilized and transported to streams during future storms, owing to the inter-connectivity of skid trails and roads with the stream system (Sidle et al., 2004b). Throughout steeplands of Southeast Asia, these hydrogeomorphic pathways are important linkages of sediment to streams. Thus, small road-related landslides (1–20 m^3) cannot be ignored as sediment sources in tropical catchments as has often been the case.

Fig. 8. Road-related landslides in Butik Tarek Catchment C3, Malaysia, associated with recent skid trail construction and disturbance along the logging road, late 2003 to early 2004, during residual stand clearance and plantation establishment.

connection of paths to channels; and (4) smooth surfaces of paths. Similar scenarios have been reported to increase sediment delivery to headwater channels in tropical catchments with systems of forest roads and skid trails (Sidle et al., 2004b). Thus, consideration of the spatial distribution of the road and path network in a catchment is potentially more important than the effects of land cover, although it is acknowledged that the two are interrelated.

4.5.2. Landslides

Field investigations worldwide have shown that landslide erosion from forest road right-of-ways is on average 25–350 times higher than in undisturbed forests (e.g., O'Loughlin, 1972; Morrison, 1975; O'Loughlin and Pearce, 1976; Gray and Megahan, 1981; Amaranthus et al., 1985). Estimated average landslide erosion rates from forest roads in steep terrain range from 3.5 to 334 t ha^{-1} yr^{-1} with most values in the range 20–45 t ha^{-1} yr^{-1}. These values are similar to surface erosion losses from forested hillslopes that have been converted to agricultural crops or plantations in Southeast Asia. One of the few comprehensive surveys of road-related landslides in the tropics noted that landslide-affected areas around roads were five to eight times higher than areas outside of the road influence (Larsen and Parks, 1997). This study in Puerto Rico was based on aerial photo interpretation and thus did not account for many smaller landslides that would occur (e.g., Brardinoni et al., 2003). Results of a field survey of small landslides associated with forest road disturbance and skid trail construction in Peninsular Malaysia are shown in Example 7.

Landslides related to roads occur episodically during years with large storms, but they can occur any time after road construction (Sidle et al., 1985; Douglas et al., 1999; Chappell et al., 2004). Alterations in hillslope hydrology by roads affect slope stability. Road drainage that is either intentionally or unintentionally directed onto portions of the hillslope has the potential to increase pore water pressure in soils during peak rainfall events (Megahan, 1972, 1983; Douglas et al., 1999). Even with well-constructed roads, too few or poorly placed drainage relief culverts can lead to pore water pressure accretion in fill material or downslope sites (Burroughs, 1984; Piehl et al., 1988). Particular areas of concern are geomorphic hollows or hillslope depressions that accumulate subsurface water and are sites of previous mass wasting (e.g., Sidle, 1984; Tsukamoto and Ohta, 1988). Storm drainage routed into these naturally wet and unstable hollows as well as roads built across them are major causes of landslides (e.g., LaHusen, 1984). In steep terrain, mid-slope roads generally have the greatest destabilizing impact by virtue of subsurface water interception and overloading and undercutting slopes (e.g., Skaugset et al., 1996; Douglas et al., 1999); however, Montgomery (1994) notes that drainage concentration from ridgetop roads can also contribute to landslide initiation. Thus, it is apparent that any road drainage that concentrates on steep slopes, in hollows, or on the road prism itself will dramatically increase the probability of slope failure. In areas where roads are built into steep hillslopes without adequate drainage, the potential for landslides is very high (e.g., Bansal and Mathur, 1976; Haigh, 1984; Thakur, 1996; Douglas et al., 1999). Hollow log culverts used as cross-drains decay with time and collapse, potentially leading to landslide initiation (Douglas et al., 1999).

While roads are a necessary part of most forest and upland agricultural land uses, the critical concerns related to slope stability are: the length of roads in steep terrain; cutting roads at mid-slope locations (especially considering the width of the road); interception and removal of water in the road right-of-way (including drainage design); recognition of highly unstable landscape features (e.g., geomorphic hollows, old slump blocks); overall road design, layout, and construction considerations; maintenance; and ultimate life and use of the road (Sidle et al., 1985; Megahan, 1987; Allison et al., 2004). Compared to roads, small trails on steep hillsides have much less impact on slope stability since little excavation is generally required.

5. Summary and conclusions

Herein, we have addressed a broad set of forest land uses and management schemes in terms of their effects on both surface and landslide erosion in Southeast Asia. These erosion processes and management interactions have been linked to dynamic hydrologic conditions whenever possible. While it is clear that surface erosion and landslide processes are very different and respond to different triggering mechanisms, it is nevertheless surprising that frequent misconceptions exist regarding the relative importance of these processes and the implications of management activities and avoidance/control measures. Although surface erosion in steep terrain related to various management practices has been widely investigated in Southeast Asia, almost no long-term research has been conducted on landslide erosion. While the emphasis on surface erosion related to upland disturbances in the tropics is justifiable, in part due to the relatively thin organic horizons in converted forest hillslopes, landslide erosion cannot be ignored in steep terrain (e.g., Harper, 1993; Phien-Wej et al., 1993). This discrepancy reflects the short-term and small-scale nature of many studies (e.g., Presbitero et al., 1995; Hartanto et al., 2003; van Dijk and Bruijnzeel, 2004) as well as the emphasis on agronomic production at the farm or field-plot scale (e.g., Craswell et al., 1998; Lal, 1998). Even in longer-term catchment studies, the occurrence of landslides and their influence on sediment budgets has been studied only in a few cases (Douglas et al., 1999; Chappell et al., 2004). Thus, due to their episodic nature, landslides processes are typically ignored or merely mentioned in passing, but they are not often quantified and have not been thoroughly investigated in Southeast Asia. To fill this void, landslide losses must be quantified via long-term sediment budgets to adequately characterize erosion and sedimentation processes. Such information is crucial in formulating best management forest practices, developing models, and dispelling myths about landslide processes in the region. For example, it is necessary for researchers, land managers, and donor groups to recognize that landslide probability does not increase substantially until several years after forest conversion and may then persist indefinitely if weaker rooted crops or plantations continue. Thus, studies and monitoring must be designed to accommodate such temporal dimensions if landslide erosion is to be quantified.

It is clear that different planning and control measures are necessary to ameliorate the effects of land management on surface erosion and landslides. For example, many of the water control practices used along roads to minimize surface erosion may not be effective measures for landslide prevention. All control measures need to focus on how hydrologic pathways will be altered by different land uses. Such effects include concentration of road drainage onto unstable slopes, gully incision on trails and road surfaces, changes in soil water due to vegetation conversion, and rerouting subsurface water to overland flow in lands converted to agricultural, along road cuts, and on terraces. In addition to reducing root cohesion, conversion of forest vegetation on steep hillsides to crops, grasslands, or plantations which require tillage will alter soil moisture regimes and hydrologic pathways, thereby increasing the probability of landslide occurrence on-site or downslope. Surface erosion control should focus on maintaining ground cover and minimizing surface runoff (and velocity), while landslide prevention must focus on minimizing pore water pressure accretion on steep, unstable slopes.

The results summarized here show that the highest levels of soil loss per unit impacted area generally come from roads and trails, but losses from converted forest lands are high in mountainous areas if sites are heavily disturbed by cultivation and when crops or plantations have poor overall ground cover. Most of these reported soil losses do not include estimates of landslide erosion, which over the long-term, may contribute comparable levels of sediment in steep terrain. The high levels of soil loss are partly related to soil and site characteristics, and partly to the high intensity of monsoon storms in Southeast Asia. Poorly constructed and maintained terraces, especially when wet, are not effective in controlling field-scale erosion; additionally, other 'linear' surface erosion control strategies (contour-hedgerows, alley cropping) may not effectively ameliorate surface erosion for all soil types. More passive conservation practices such as, reduced tillage, contour tillage and planting, maintaining a viable ground cover, and strip cropping may be more effective erosion control measures. Furthermore, terrace construction can contribute to landsliding by impounding water and oversteepening terrace faces (risers). In contrast to the relatively high erosion rates on converted lands, soil losses after timber harvesting (with subsequent forest regeneration) are relatively low. Much of the erosion attributed to forest operations is typically from forest roads (including landslides), skid trails, and log landings—sites of severe compaction and disturbance; forest harvesting alone (felling and yarding of trees) generally has a minor affect on surface erosion unless severe and widespread disturbance occurs. Clearcutting in steep terrain has been shown to increase landslide erosion by 2–10-fold based on studies conducted in North America (summarized by Sidle et al. (1985)). Thus, similar processes that contribute to this increased landslide susceptibility in temperate regions (i.e., reduced rooting strength during the 3–15 yr window after harvesting), as well as effects of increases in soil moisture and decreases in secondary permeability after harvesting, may cause tropical forests to be even more sensitive to landslide erosion after timber harvesting. The effects of recreation activities in Southeast Asia on erosion processes are a conspicuous omission in this paper. While widespread land cover changes in support of recreation have been noted to increase erosion and sedimentation in the region (e.g., Gupta, 1993; Wan Ruslan, 1997; Lu et al., 2001), few studies have attempted to link either surface erosion or landslide processes with specific recreational impacts, such as forest clearance, resort construction, water diversions, roads, hiking and animal trails, and all-terrain vehicle tracks, in steep terrain.

Few of the studies that were cited attempted to clarify the effects of scale (related to distributed land uses) on sediment delivery to streams. Surface erosion rates measured at different scales in converted catchments *will not* be the same, even if land use and slope gradients are similar. Plot scale erosion is generally higher than catchment scale erosion because average infiltration capacity increases with scale and deposition occurs within the catchment (van de Giesen et al., 2000). However, in some cases, larger scale measurements will include other processes (e.g., gully development, bank erosion, dry ravel) that cannot be captured at the plot scale, and thus may yield higher erosion losses. If total sediment yields are considered (i.e., including landslides), the scale issue is further complicated. Scaling of surface erosion and landslides is a fundamental problem and research priority in the region that can only partly be solved by modeling—additional field studies are needed (e.g., nested catchment studies) to assess the various erosion processes at different spatial and temporal scales. However, we do know that a continuity of land disturbance from up- to downslope will facilitate sediment transport to streams. Roads, trails and footpaths within these land management mosaics will further exacerbate linkages to streams. Fragmented intensive land uses that are interspersed by trees or brush land appear to be a viable solution for mitigating downslope sediment transport by providing areas of high infiltration along with 'roughness elements' on the landscape where sediment deposition can occur. A better understanding of sediment transport and routing processes is needed at the catchment scale to develop improved predictive methods and to assess the cumulative effects of distributed tropical land uses.

Recognition of the "truths, myths, and uncertainties" related to erosion processes and consequences in tropical Southeast Asia will assist land managers, land owners, and policy makers in formulating appropriate and prudent decisions that will contribute to more sustainable use of forest lands as well as options for rehabilitation of previously forested lands that have been degraded. Benefits derived from improved land management practices will not only be accrued by land owners (e.g., increased land productivity, reduced soil loss), but also by off-site and downstream residents, due to improved water quality, decreased sedimentation, diminished natural hazards, and enhanced fisheries and aquatic habitat. Nevertheless, international agencies and local governments need to be more responsive to the research needs outlined herein, as these issues rest at the heart of linking land management to sustainable strategies for land and water resources as well as food security in the region.

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