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Land use impacts on water resources: A literature review

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The present discussion paper summarizes the findings of a desk study on the impacts of land use on water resources. It has been prepared as a reference and basis for debate in the first part of the e-workshop on land-water linkages in rural watersheds.

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INTRODUCTION

Land use practices are assumed to have important impacts on both the availability and quality of water resources. These impacts can be both positive and negative. It is intuitively appealing that the benefits of improved land management, or the costs associated with negative impacts of inadequate land use on the water resources, might not only be felt by water users who cause them, but also by others who live downstream or – in the case of groundwater – make use of the affected groundwater resources. In order to assess these costs and benefits, it is important to get a clear picture, from a landscape perspective, of the extent that different land use practices affect hydrologic regime and water quality and at which watershed scale the impacts are of importance.

The present paper proposes a typology of land use impacts on water resources, and attempts to evaluate the importance of each impact type in relation to spatial scale on the basis of a literature review.

IMPACTS OF LAND USE ON WATER RESOURCES

In order to establish linkages between upstream land and downstream water users, it is important to have a clear picture of the possible impacts of land uses on both hydrologic regime (water availability) and water quality, and the scales at which these impacts are relevant. In the following sections, an attempt is made to categorise land use impacts on water resources, to analyse the main determining factors behind the impact, and to provide some examples from the literature.

The review focuses on impacts from agricultural land use, as well as from grazing, forestry and fisheries, as these all fall under FAO's mandate. Other land uses, like mining and quarrying, urbanization and industrialization, which also have important impacts on the hydrologic regime, are not included in this review. Furthermore, the review focuses on the physical impacts on water resources. Impacts on living aquatic resources, e.g. on fish and other aquatic organisms, aquatic ecosystems and wetlands, are not discussed explicitly. It is an open question, however, whether and how these should be included in this typology.

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It is difficult to arrive at universally valid statements about land use impacts on water resources for several reasons. The impacts of land use on water resources depend on a host of natural and socio-economic factors. Natural factors include climate, topography and soil structure. Socio-economic factors include economic ability and awareness of the farmers, management practices, and the development of infrastructure, e.g. roads. Furthermore, the impacts of agricultural land use may be difficult to distinguish from natural or other human impacts, such as the impact of agricultural runoff versus rural sewage systems on degradation of surface water and groundwater.

Land use impacts on hydrologic regime

With regard to the hydrologic regime, impacts on surface water resources and groundwater resources can be distinguished. Impacts of land use practices on surface water can be divided into (i) impacts on the overall water availability or the mean annual runoff, and (ii) impacts on the seasonal distribution of water availability. With regard to the latter, impacts on peak flows and impacts on dry season flows are of importance. With regard to groundwater, the effect of the land use on groundwater recharge has to be examined.

Mean surface runoff

The impact of land use on the mean runoff is a function of many variables, the most important being the water regime of the plant cover in terms of evapotranspiration (ET), the ability of the soil to hold water (infiltration capacity), and the ability of the plant cover to intercept moisture.

A change of land cover from lower to higher ET will lead to a decrease in annual stream flow. From a review of 94 catchment experiments, Bosch and Hewlett (1982) concluded that the establishment of forest cover on sparsely vegetated land decreases water yield. Coniferous forest, deciduous hardwood, brush and grass cover have (in that order) a decreasing influence on water yield of the source areas in which the covers are manipulated.

Conversely, a change from higher-ET plants to lower-ET plants will increase the mean surface runoff: reduction in forest cover increases water yield (Bosch and Hewlett, 1982; Calder, 1992). The impact, however, depends very much of the management practices and the alternative land uses. Careful, selective harvesting of timber has no or little effect on stream flow. Stream flow after maturation of the new plant cover may be higher, the same or lower than original value, depending on vegetation (Bruijnzeel, 1990).

An exception to this rule are cloud forests, which can intercept more moisture (fog drip) than consumption by ET (Bosch and Hewlett, 1982), and very old forests, which, depending on the species, may consume less water than the vegetation that establishes itself after clear-cutting (Calder, 1998).

Stream flow gains decline over time with establishment of new plant cover, but time scales can vary greatly. In humid warm areas, the effect of clear-cutting is shorter lived than in less humid areas, due to rapid regrowth of vegetation (Falkenmark and Chapman, 1989).

Increasing water yield from changing plant cover does not necessarily increase water availability downstream. Stream flow might decrease because of other factors, e.g. water consumption by riparian vegetation or through transmission losses (channel infiltration) (Brooks *et al.*, 1991).

Peak flow/floods

Peak flows can increase as a result of a change in land use if the infiltration capacity of the soil is reduced, for example through soil compaction or erosion, or if drainage capacity is increased. Peak flow may increase after trees are cut down (Bruijnzeel, 1990). Relative increases in storm flow after tree removal are smallest for large events and largest for small events. As the amount of precipitation increases, influence on storm flow of soil and plant cover diminishes (Bruijnzeel, 1990; Brooks *et al*., 1989).

An increase of peak flows may also result from the building of roads and infrastructure. Studies in the north-western USA have shown that the construction of forest roads can intensify peak runoff from forested areas significantly (La Marche and Lettenmaier, 1998; Bowling and Lettenmaier, 1997). Consolidation of smaller plots to large fields can lead to higher runoff rates, due to drainage systems and asphalt access roads (Falkenmark and Chapin, 1989). Conversely, peak flows may decrease as a result of an increased soil infiltration capacity.

In larger basins, effects of land use practices on peak flow are offset due to time lag between different tributaries, different land use and variations in rainfall (Bruijnzeel, 1990) In larger watersheds, this de-synchronisation effect can lead to a reduction in peak discharge, although overall storm flow increases due to land use changes in individual subwatersheds (Brooks *et al*., 1991).

Base flow/dry season flow

The effect of land use change on dry-season flow depends on competing processes, most notably changes in ET and infiltration capacity. The net impact is likely to be highly site specific (Calder, 1998).

In tropical areas, afforestation can lead to deceased dry-season flows due to increased evapotranspiration. In the Mae Thang watershed (Thailand), afforestation programmes led to water shortages downstream, which resulted in a seasonal closure of a water treatment plant and lower availability for irrigation (Chomitz and Kumari, 1996). Similarly, in the Fiji Islands, large-scale pine afforestation (60 000 ha) in watersheds previously covered by grassland led to reductions in dry-season flow of 50-60 percent, putting the operation of a hydro-electric plant and drinking water supply at risk (FAO, 1987).

Most experimental evidence in rainfall-dominated regimes suggests that forest removal (or change from high-water-use plants to low-water-use plants) increases dry season flows (Brooks *et al*., 1991). In contrast, dry-season flows from deforested land may decrease if the soil infiltration capacity is reduced, e.g. through use of heavy machinery (Bruijnzeel, 1990). Low flow resulting from extended dry periods or droughts may not be substantially altered by changes in vegetative cover (Brooks *et al*., 1991).

Groundwater recharge

The groundwater recharge may be increased or decreased as a result of changing land use practices. The major driving forces are the ET of the vegetative cover and the infiltration capacity of the soil. Groundwater recharge is often linked with dry-season flows, as groundwater contributes much of the river discharge during the dry season.

The water table may rise as a result of decreased evapotranspiration, e.g. following logging or conversion of forest to grassland for grazing. Recharge may also increase due to an increased infiltration rate, e.g. through afforestation of degraded areas (Tejwani, 1993).

In contrast, the water table may fall as a result of decreased soil infiltration, e.g. through non-conservation farming techniques and compaction (Tejwani, 1993). Also, heavy grazing may lead to reduced infiltration and groundwater recharge (Chomitz and Kumari, 1996). If the infiltration capacity is substantially reduced, this can lead to water shortages in dry seasons, even in regions where water is usually abundant, like in the case of shifting cultivation in Cherapunji province, India (FAO, 1999). Likewise, groundwater recharge can be reduced as a result of planting of deep rooting tree species, e.g. eucalyptus (Calder, 1998).

Impacts of land use on water quality

Land use practices can have important impacts on water quality, which in turn may have negative or, in some cases, positive effects on downstream uses of water. Impacts include changes in sediment load and concentrations of nutrients, salts, metals and agrochemicals, the influx of pathogens, and a change in the temperature regime.

Erosion and sediment load

Forests are checkers of soil erosion. Protection is largely because of understorey vegetation and litter, and the stabilising effect of the root network. On steep slopes, the net stabilising effect of trees is usually positive. Vegetation cover can prevent the occurrence of shallow landslides (Bruijnzeel, 1990). However, large landslides on steep terrain are not influenced appreciably by vegetation cover. These large slides may contribute the bulk of the sediment, as for example in the middle hills of the Himalayas (Bruijnzeel and Bremmer, 1989).

Afforestation does not necessarily decrease soil erosion. Splash erosion may increase substantially when litter is cleared from the forest floor (Bruijnzeel, 1990). The spectrum for the size of the drops that are formed by the canopy varies widely among different species, resulting in large differences in the potential of splash erosion (Calder, 1998).

Deforestation may increase erosion. In Malaysia, streams from logged areas carry 8-17 times more sediment load than before logging (Falkenmark and Chapman, 1989). The actual soil loss, however, depends largely on the use to which the land is put after the trees have been cleared. Surface erosion from well-kept grassland, moderately grazed forests and soil-conserving agriculture are low to moderate (Bruijnzeel, 1990).

Road construction may be a major cause for erosion during timber harvesting operations. In the USA, forest roads are estimated to account for 90 percent of the erosion caused by logging activities (Brooks *et al*., 1991; Bruijnzeel, 1990).

Effects of erosion control measures on sediment yield will be most readily felt on-site. There is an inverse relation between basin size and sediment delivery ratio. In basins of several hundred km^2, improvements may only be noticeable after a considerable time lag (decades), due to storage effects (Bruijnzeel, 1990).

Downstream sediment yields cannot always be ascribed to the changing of upstream land use practices. Human impacts on sediment yield may be substantial in regions with stable geological conditions and low natural erosion rates. In regions with high rainfall rates, steep terrain, and high natural erosion rates, however, the impact of land use may be negligible. In the Phewa Tal watershed in Nepal, for example, only six percent of the total sediment yield has been calculated to stem from surface erosion (Bruijnzeel, 1990).

Sediment can act both as a physical and a chemical pollutant. *Physical pollution* characteristics of sediment include turbidity (limited penetration of sunlight) and sedimentation (loss of downstream reservoir capacity, destruction of coral reefs, loss of spawning grounds for certain fish). *Chemical pollution* of sediment includes adsorbed metals and phosphorous, as well as hydrophobic organic chemicals (FAO, 1996).

Nutrients and organic matter

A change in land use can alter the nutrient content of surface and groundwater, most notably nitrogen (N) and phosphorus (P) levels. Deforestation can lead to high nitrate (NO_3) concentrations in water due to decomposition of plant material and a reduced nutrient uptake by the vegetation. Nitrate concentration in runoff in deforested catchments can be 50 times higher than in a forested control catchment over several years (Falkenmark and Chapman, 1989; Brooks *et al*., 1991).

Agricultural activities can lead to an increased influx of nitrogen into waterbodies as a result of many factors, including fertiliser application, manure from livestock production, sludge from municipal sewage treatment plants, and aeration of the soil. In Europe, agriculture accounts for substantial nitrogen emissions into surface and groundwater. With regard to inorganic N, agriculture accounts for 50 percent in Denmark and 71 percent in the Netherlands (FAO, 1996). High nutrient leaching losses can occur when fertiliser is applied to short-term crops on permeable soils. In Sri Lanka, NO_3-N concentration in groundwater under intensive chilli and onion cultivation reaches 20-50 mg/L (BGS *et al.*, 1996). Continuous soil cover reduces N leaching; fallow periods and soil disturbance increases leaching (BGS *et al.*, 1996). Ploughing can increase NO_3 concentrations in surface and groundwater, as oxygenation of the soil causes nitrification (Falkenmark and Chapman, 1989). In rice paddies, leaching losses are likely to be small, due to denitrification in the soil and volatile losses (BGS *et al.*, 1996). Application of manure from livestock production and direct runoff can lead to acidification of soils due to the volatilisation of ammonia, which in turn may increase the solubility of metals in the soil (FAO, 1996).

Phosphate (PO_4) leaching into water is inhibited by sorption processes to clay particles (BGS *et al.*, 1996). Livestock production, however, can be a major source of P in waters. Direct runoff from intensive livestock farms can lead to serious degradation of surface and groundwater. In the EU, livestock wastes account for 30 percent of P load in surface waters, other agricultural uses account for 16 percent (FAO, 1996).

Phosphate-laden sediment can form a nutrient pool on the bottom of eutrophic lakes, which can be released into the water under anoxic conditions. This makes it difficult to control eutrophication in the short term through limitation of P inflow. Eutrophication can be mitigated by dredging sediment or oxidising the hypolimnion, but these options are quite costly (FAO, 1996).

The precise role of agriculture in the contamination of ground and surface water is difficult to quantify. In most countries, monitoring is not sufficient to establish the extent of nutrient pollution from agricultural land use. In rural areas, it may be difficult to distinguish between agricultural pollution and pollution by untreated sewage (BGS *et al.*, 1996).

Freshwater aquaculture can add substantial nutrient loading to surface water through waste feed that is not consumed by the fish, and the fish's faecal production (FAO, 1996).

Pathogens

Land use activities may affect the bacteriological quality of water, which can create health concerns for downstream water users. The concentration of pathogenic bacteria in surface waters may increase as a consequence of riparian grazing activities or waste influx from livestock production.

A reduction of stream flow, for example, as a consequence of upstream diversion for irrigation, may lead to ponding in riverbeds, which in turn may provide breeding grounds for vectors of waterborne diseases, such as malaria. Where low flow leads to saltwater intrusion in estuaries, vectors breeding in brackish water may spread (FAO, 1995).

Pesticides and other persistent organic pollutants

Generally, the application of pesticides poses a danger to surface and groundwater resources, since pesticide compounds are designed to be toxic and persistent. Pesticide leaching into groundwater depends on the chemical's persistence and mobility, as well as the soil structure. Pesticide metabolites might be as toxic and as mobile as the parent compound (BGS *et al.*, 1996). In humans and animals, pesticides can have both acute and chronic toxic effects. Lipophilic compounds can accumulate in fatty tissue (bio-concentration) and in the food chain (bio-magnification) (FAO, 1996).

Pesticide residues can find their way into water resources through their use in agriculture, forestry and aquaculture. Furthermore, unsafe stockpiling and dumping of old and obsolete pesticides can cause severe ground and surface water contamination (FAO, 1996). Aquaculture can lead to the introduction of biocides, disinfectants and medicines into surface water (FAO, 1996).

The actual impact of pesticide contamination of downstream water resources is often difficult to quantify. Pesticide monitoring is difficult because concentrations are very low, large samples and careful sampling, as well as sophisticated analytical instruments, are required (BGS *et al.*, 1996). Since many pesticides are transported in association with suspended matter, water analyses may yield incomplete results. For some pesticides, the analytical capability may not be accurate enough to determine presence or absence for the protection of human health. Newer pesticides which are soluble and degrade more quickly can only be detected shortly after application; therefore, typical monitoring programmes operated on a monthly or quarterly basis are unlikely to be able to quantify the presence and determine the significance of pesticides in surface waters (FAO, 1996).

Salinity

An increase in salinity of surface and groundwater can have detrimental effects on downstream water uses, for example for irrigation or domestic water supply. The impact of land uses on salinity depends on climatic as well as geological factors.

Irrigation and drainage activities may lead to increased salinity of surface and groundwater as a consequence of evaporation and the leaching of salts from soils. This is of special concern in arid areas, where subsurface drainage water always has higher salt concentrations, an increased hardness and a higher sodium absorption ratio than the supply water (FAO, 1997a). Drainage from irrigated agriculture may also lead to an increased concentration of selenium in ground and surface water (Postel, 1997).

A high application rate of potassium chloride fertiliser can lead to an increased leaching of chloride into groundwater. In Sri Lanka, for example, it has been estimated that in some areas of intensive agriculture, groundwater chloride levels may rise to 400 mg/L by 2010 at current rates of fertiliser application, which by far exceeds the acceptable concentration for drinking water as determined by WHO (250 mg/L) (BGS *et al.*, 1996).

In coastal areas, water abstraction for land use activities may indirectly contribute to the salinization of water resources. Groundwater extraction for irrigation, domestic and industrial purposes can result in the intrusion of seawater into the aquifer, and consequently a salinization of the groundwater resources (FAO, 1997). A decrease in river flow due to upstream abstraction or the building of reservoirs can lead to an inland intrusion of brackish water in the estuarine zone (FAO, 1995).

Heavy metals

Land use practices may directly and indirectly contribute to an increased concentration of heavy metals in water resources. A direct pathway is the application of livestock manure and sludge from sewage treatment plants, which may contain high concentrations of heavy metals. For example, pig manure often contains high concentrations of copper (FAO, 1996).

Indirectly, land use may affect heavy metal concentration in surface and groundwater by increasing the mobility of metals in the soil from anthropic or geological origin. Heavy metals in the soil may be transferred into waterbodies by erosive processes. The acidification of soil, caused by ammonia volatilization from manure application or in animal feedlots, may increase the solubility of heavy metals stored in the soil, and thus the influx into surface and groundwater. High abstraction rates of groundwater for irrigation can alter the chemical environment in the soil, leading to an increased mobility of heavy metals of geological origin. This may be the reason for increased arsenic concentration in Bangladesh (Ahmed and Amin, n.d.).

Changes in thermal regime

The thermal regime of surface water can be affected by land use practices. In small streams, removal of riparian vegetation can cause temperature increase in the water (thermal pollution) (Brooks *et al.*, 1991). Also, tail water discharge from irrigated areas may cause a rise in temperature of the receiving stream (FAO, 1997a). A temperature rise leads to reduced oxygen solubility, which can negatively affect the biological activity in the water as well as the self-cleaning capacity of the river.

Scale considerations

The above review of land use impacts on water resources does not take into account spatial and temporal distribution aspects. Scale considerations, however, are of fundamental importance when assessing these impacts as they indicate whether a land use upstream may affect a water use downstream.

Spatial scale

With regard to the spatial scale, i.e. the size of river basin, the land use impact can become less important because of offset effects, such as de-synchronisation (e.g. in the case of floods), storage capacity of the river bed (sedimentation) or the self-cleaning capacity of the river (organic pollution). At the same time, the impact can become more important with increasing scale due to accumulative effects, e.g. in the case of salinity.

Land use induced changes of the hydrologic regime and sediment load decrease with the size of the river basin. The effects will be most readily felt in smaller watersheds of up to several hundred km^2. One well-documented case is the Ganges-Brahmaputra-Meghna basin. Studies show that in small-scale catchments (<50 km^2) in the basin, erosion and stream flow may be strongly influenced by changing land use patterns (Ives and Messerli, 1989). However, the lowland flooding in Bangladesh is not related to the increased peak flow and erosion resulting from deforestation in the Himalayan uplands in Nepal. The main driving forces behind the flood events in the plains are naturally occurring rainfall events in the lowlands, which may be augmented by human interventions in the floodplains, such as road and river embankments (Hofer, 1998a; Ives and Messerli, 1989). Similarly, the bulk of the sediment load in the Ganges-Brahmaputra river system does not stem from human-induced erosion, but rather from large landslides not influenced by human activity (Bruijnzeel and Bremmer, 1989).

With regard to water quality impacts, the picture is much less clear. Observations show that some land use impacts on water quality, like salinity or pesticide load, can also have downstream effects in medium to large watersheds, like the Murray-Darling basin (Australia) and the Colorado basin (USA/Mexico). Other downstream impacts, like organic matter and pathogens, are relevant only at smaller scales.

The spatial dimensions of land use effects can be summarized as follows:

Impact	Basin size [km^2]						
	0.1	1	10	100	1 000	10 000	100 000
Average flow	x	x	x	x	-	-	-
Peak flow	x	x	x	x	-	-	-
Base flow	x	x	x	x	-	-	-
Groundwater recharge	x	x	x	x	-	-	-
Sediment load	x	x	x	x	-	-	-
Nutrients	x	x	x	x	x	-	-
Organic matter	x	x	x	x	-	-	-
Pathogens	x	x	x	-	-	-	-
Salinity	x	x	x	x	x	x	x
Pesticides	x	x	x	x	x	x	x
Heavy metals	x	x	x	x	x	x	x
Thermal regime	x	x	-	-	-	-	-

Legend: x = Obervable impact; - = no observable impact

Temporal scale

Temporal scale is another important aspect of land use impacts, as it determines the perception of the impact as well as the economic cost associated with it. Two aspects are important with regard to temporal scale of land use impacts. First, the time it takes for a land use to have an impact on downstream uses, and, second, in the case of negative impacts, the time it takes for remedial measures to take effect, if the impact is reversible.

The temporal scales of land use impacts vary widely. Depending on the impact, they may range from less than one year, as in the case of bacterial contamination, to hundreds of years, as in the case of salinity. Similarly, time scales of recovery from adverse impacts are very diverse, depending on the impact. However, in most cases, the time it takes to restore an aquatic system after an adverse impact is much longer than the time it takes for an impact to appear (Peters and Meybeck, 2000).

Conclusion

With regard to land use impacts on hydrologic regimes and sediment transport, there is an inverse relationship between the spatial scale in which the impacts can be observed and the scale in which the redistribution of benefits might be important. These impacts can be most readily felt in small spatial scales. At the same time, the number of water users who might benefit or suffer from this land use change, increases with the size of the watershed. Due to the decreasing magnitude of impact, the respective costs and benefits will be small. Impacts of land use practices on water quality, like salinity, pesticide pollution and eutrophication due to nutrient influx, however, may be relevant in medium- to large-scale river basins as well. These impacts may affect many downstream uses, including providers of drinking water, industries, fisheries and other agricultural uses.

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