Chapter 13
Land Use

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> . . .In every respect the valley rules the stream. [. . .] We must, in fact, not divorce the stream from its valley in our thoughts at any time. (Hynes 1975)

13.1 Introduction

Dendritic stream-river networks are the backbone of most landscapes on earth's surface and determine linkages between terrestrial and aquatic ecosystems. These networks are hierarchically organized from microhabitats to the scale of whole catchments (Frissell et al. 1986). Accordingly, many processes of lotic freshwater ecosystems are determined by this hierarchically nested structure of river networks and their interlinkages. Hynes (1975) emphasized the importance of the linkage between the conditions within a river catchment and the flows of energy, materials, and organisms in the river as a dynamic ecosystem. These flows are interwoven in complex, cross-linked relationships of ecosystem functioning. Up to now, it is a prime challenge to understand these functions in detail and to robustly manage riverine landscapes in a sustainable manner. Starting with the work of Hynes (1975), the scale of riverine management was understood to occur over entire river catchments or even river basins with several concepts that integrated this perception (Vannote et al. 1980; Frissell et al. 1986; Ward 1989; Poff 1997; Fausch et al. 2002; Ward et al. 2002; Burcher et al. 2007). These theoretical frameworks seek to understand and to quantify interactions between landscape conditions over large spatial extents and instream responses. Ultimately, the catchment approach was even implemented into legal frameworks, such as the Water Framework Directive (WFD), which recognizes the river basin as relevant management scale (European Parliament 2000).

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However, the bases of these concepts originate in landscape ecology and its inherent landscape-scale thinking, which was traditionally focused on terrestrial ecosystems. In turn, relationships between landscape patterns and their consequences are dependent on the characteristics of the mosaic of the surrounding landscape at multiple temporal and spatial scales. Moreover, river networks are highly effective in linking different landscape elements even over large distances. Hence, Wiens (2002) suggested to take the "land" out of landscape ecology to emphasize the importance of integrating landscape ecological approach into river research and to deepen the understanding on the interplay between terrestrial and aquatic ecosystems which should enable to perceive riverine ecosystems as "riverscapes" as proposed by Ward (1998). After Fausch et al. (2002) and Wiens (2002), encouraging researchers and managers to consider the entire river environment, it was Burcher et al. (2007) who introduced the *land cover cascade* concept in which disturbance stimuli are propagated through a series of hierarchical entities until they ultimately affect biota in their habitat. Subsequently, researchers expanded their thinking to integrate this "bigger picture" into pressure-impact analyses. Several studies statistically linked landscape characteristics, that describe the conditions over large spatial extents, to biological and environmental indicators of ecosystem states.

This chapter emphasizes the importance of the catchment scale in river ecology and how human actions in the landscape with related multiple alterations result in pressures on riverine ecosystems impacting habitat quality and aquatic biota. At the beginning, land use terminology will be explained, followed by methods and data that are used to describe, detect, and investigate landscape-scale patterns in relation to lotic ecosystems. Subsequently, the linkage of land uses and pressures will be highlighted followed by the characterization of possible management actions to mitigate negative human-induced land use effects.

13.2 Land Use and Land Cover Definitions

Land use in river catchments is widely understood to have an effect on terrestrial but also aquatic ecosystems and especially on their interlinkage. Humans utilize land area with the intention to obtain products and/or benefits by using resources and—in most cases—changing the properties of land patches. Any human action and usually a series of operations on land define a type of land use. Hence, the term *land use* refers to the purpose the land serves, e.g., agriculture or urban areas, and it does not necessarily describe the (bio)physical materials at the earth's surface. The description of vegetation, materials, objects, and bare surface, either natural or man-made, commonly defines *land cover* types (Fisher et al. 2005).

These two terms, land use and land cover, are often erroneously used interchangeably, although each term has a very specific meaning as mentioned above. They are fundamentally distinct in their genesis, purpose, and application. Land cover is the direct observation, mostly from top view imagery, while land use requires additional socio-economic interpretation of the activities that take place on that surface (Comber 2008a).

Land use gives answers to the “why?” people need, use, and change the land, whereas land cover is important to the “how?” are ecosystems affected when inferring process-based fluxes of energy and material. Therefore, land use and land cover are distinct concepts in earth surface modeling approaches. The research community requires land cover for environmental models and land use for policy making (Comber 2008b).

There is no one-to-one relationship between land use and land cover types. The land cover “grass” can be found in multiple land uses like urban parks, residential land, pasture, etc. On the other hand, many homogeneous land use types encompass more than one land cover; residential land may contain trees, grass, buildings, and paved surfaces. Land use for recreation describes possible activities but can be applicable to different land cover types: the surface for recreational activities can, for instance, be a sandy beach, a built-up area like a pleasure park, woodlands, or even a pond. Hierarchical classification of land use is one approach to integrate cover information and human purpose in one dataset. Lower levels of land use classes are likely to have a one-to-one land cover equivalent.

In planning and decision-making processes, both the land use and land cover of a particular area (e.g., a river catchment) provide a comprehensive picture of the situation. Depending on the processes intended to be studied, and depending on the available data, it is crucial to select and prepare land use datasets with thematic and spatial resolution that is appropriate for the spatial extent of the study.

In communication between disciplines, research and practice, management, and conservation, this distinction between land use and land cover should be made clear. Such data on land use and land cover are often used indifferently in practice, leading to thematic inconsistencies in mapping and classification that can cause problems of further interpretation (Loveland et al. 2005).

In riverine ecosystem research and management, land use is seen as integrator of human actions in the landscape (Allan and Castillo 2007). For certain purposes, land use evidentially affects the ecological integrity of river ecosystems by altering spatially nested controlling factors of different scales (Allan 2004). Many, although not all, impacts on streams are entirely or partly linked to human actions in the landscape and thus can be quantified from data on land use.

Datasets on land use are often preferred over land cover datasets because the latter, emphasizing vegetation cover, is subject to seasonal changes and geographic particularities (latitude, biogeographical regions, etc.), making a universal classification scheme difficult to realize.

13.3 Methods and Data in Land Use Analysis

Current land use and land cover data are most often obtained from analysis of either satellite or aerial images. New technologies both in computer science and remote sensing provided the data and imagery that accelerated the incorporation of landscape thinking into riverine research (Johnson and Gage 1997). Before large-scale datasets became available, habitat mapping was field intensive and necessarily limited to discrete reaches. Research methods in freshwater ecology have historically been applied on individual stream reaches that were placed in their landscape context by conceptual models.

Since the 1940s, aerial photography has provided a landscape-scale perspective, but a suite of recent new technologies has dramatically improved our ability to conduct research over large areas (Steel et al. 2010). For decades, remote sensing has enabled synoptic views of entire rivers and their catchments at increasingly fine resolutions. It can now provide data at resolutions that capture reach-scale riparian and instream habitat structuring (Hall et al. 2009).

The era of satellite images for environmental monitoring started in 1972 with the launch of the Landsat program in a joint mission of the US Geological Survey (USGS) and the National Aeronautics and Space Administration (NASA). Since then satellites have continuously acquired multispectral (visible and infrared light) images of the entire earth's land surface (US Geological Survey 2016). This extensive data archive provides valuable resources for land use classification and allows for tracking land use changes through time.

There are an increasing number of sensing technologies used for mapping both landscape and water properties including optical sensors, light detection and ranging (LiDAR), forward-looking infrared (FLIR), and radio detection and ranging (RADAR) (Mertes 2002). These high-tech devices are deployed from multiple platforms that range from low-altitude tethered balloons to drones, to helicopters and fixed-wing aircraft up to outer space satellites, thereby producing data with sub-centimeter to multimeter grain resolution across scales from microhabitats to channel units to valleys to catchments.

Spatial land use data finally is the product of interpretation, separation, and classification of raw data based on four linked conditions: (1) *map scale* (e.g., degree of generalization, 1:50,000) and *spatial grain resolution* (e.g., 30 × 30 m pixel size), (2) minimum mapping unit, (3) the basic information used (e.g., satellite image or field mapping survey), and (4) the structure and number of items of the nomenclature.

One prominent example of a land use dataset that is often used for catchment analysis is the CORINE Land Cover dataset. The program runs since 1985 and was initiated by the European Union. The four characteristics of this product are map scale 1:100,000; based on Landsat 7 satellite images with 30 m pixel size; minimum mapping area for patches = 25 ha; and nomenclature with 3 hierarchical levels and 44 categories.

At present, most current studies rely on static maps that may represent land use with some years' asynchrony to the field-based stream data. However, remotely sensed data are likely to become more widely used in the future. Availability and accessibility of such data are greatly increasing in quantity and quality as opportunities grow to better synchronize time frames between data sources (remotely sensed, field based) (Allan 2004).

Basically, geographic information systems (GIS) enabled the analyses of landscape patterns. The most common practice is to calculate landscape composition metrics, i.e., expressing the total amount of various land use categories as percentage of the area under investigation (e.g., total catchment area or riparian corridor). Landscape composition metrics quantify the amounts of various land use/cover types without taking into account the spatial arrangement.

Besides a catchment's land use composition, a multitude of other landscape metrics can express the geometric properties of landscape elements and their relative positions and distributions (Botequilha Leitao et al. 2006). Landscape configuration describes arrangement, position, orientation, and neighborhoods of elements. Configurational metrics include patch size, patch shape, and compactness, as well as distance between patches. In this respect, composition metrics tell an incomplete story (Fahrig 2003). A watershed with 25% forest consisting of one large patch of contiguous forest provides significantly different ecological functions than a watershed that also contains 25% forest but is fragmented by small, scattered clear cuts.

Relatively fewer studies used landscape configuration to evaluate its divergent effects on landscape processes and ultimately on aquatic communities. Gergel et al. (2002) reviewed the advantages of landscape indicators for monitoring human impacts and finally concluded that simple metrics are useful as complements to other aquatic indicators. Simple metrics like proportion of the catchment or a buffer zone in different land covers and a measure of the arrangement or connectivity of natural and human-modified cover types in the riparian zone appear straightforward to interpret. In a later study, Gergel (2005) tested spatial and nonspatial landscape metrics within simulated landscapes of different percentages and arrangements of nutrient source and sink areas to understand watershed nutrient loadings. They found that a wide variety of different spatial configurations for watersheds with intermediate relative abundances of sources and sinks can lead to either very high or very low loading.

13.4 Land Use as Human Pressure and Its Impacts on Rivers

The pathways of cause-effect chains that are induced by land use act via hydromorphological and physicochemical controls in the ecosystem. Accordingly, natural factors of the landscape have also to be considered in analyses of land use effects. Geological bedrock material (e.g., siliceous vs. calcareous), soil types (e.g., soil fertility), and topography (steep vs. smooth slopes) are features of the natural terrain driving and underlying anthropogenic land use. Thus, anthropogenic and natural factors covary, because the latter influences the suitability of locations for the former such as agricultural and urban development (Feld et al. 2016). Land use can also be interlinked with natural gradients of geo-climatic factors, i.e., land use patterns over large extents, such as Europe, are strongly dependent on climatic factors (Lucero et al. 2011). Similarly, Feld et al. (2016) showed significant interactions of land use and geo-climatic factors in a comprehensive study covering eleven organism groups across several freshwater systems such as rivers, floodplains, lakes, ponds, and groundwater. In this study, different biodiversity indices were linked to land use descriptors, geo-climatic factors, as well as their interactions. The results showed consistently strong shared effects of land use and geo-climatic descriptors. Potentially, geo-climatic factors not only dominate but act in concert with land use. Hence, broad-scale studies on the role of environmental factors driving biodiversity must not overlook the shared effects of natural and anthropogenic descriptors. In turn, whenever anthropogenic and natural gradients covary, and only anthropogenic land use is assessed, the influence attributed to land use can be overestimated (Allan 2004).

The emphasis of landscape ecology on scale and their hierarchies has led riverine landscape research to use multi-scale studies in order to gain a better understanding of processes acting in a stream network (Lammert and Allan 1999; Fausch et al. 2002; Lowe et al. 2006). Many studies reported that a high proportion of forest cover in a catchment is normally associated with good stream conditions. Conversely, agricultural or urban areas in the catchment have been documented to have a negative influence on downstream river conditions. Agricultural land use degrades streams by increased diffuse inputs of fine sediments, nutrients, and pesticides, impacting riparian and stream channel habitats and altering flows. Increased urban land area can change the amounts and variety of pollutants in runoff, cause more erratic hydrology owing to increased impervious surface area, increase water temperatures owing to loss of riparian vegetation and warming of surface runoff on exposed surfaces, and degrade instream habitat structure owing to sediment inputs and channelization (Allan 2004).

Many studies have attempted to identify and measure land use effects and quantify the changes in land cover and the related effects on stream characteristics and associated biotic assemblages (Hughes et al. 2006). A schematic overview of the manifold effects of land use and the mechanisms disturbing aquatic biota are given in Fig. 13.1 by putting the relationships in the DPSIR framework. In most cases, the local, instream pressures and impacts related to land uses comprise sedimentation, nutrient enrichment, toxics, and hydromorphologic alterations (Allan 2004). Sedimentation clogs the interstitial of the river bed substrate, thus impairing spawning habitat for fish and habitat for benthic invertebrates. Furthermore, increased nutrient load can decrease levels of dissolved oxygen. Pesticides and other toxic substance like heavy metals impact vitality of biota. Finally, drainages and sealed surfaces change the natural characteristics of flood events (e.g., storm water runoff).

Most of these studies have investigated landscape-scale effects of single land use categories. For example, the proportion of agriculture or of urbanization in a river catchment is well-studied, and both are understood to have detrimental effects on instream conditions. From a global perspective, urban land use may appear insignificant, as urban areas only occupy less than 2% of the earth's land surface. Besides the extreme land cover changes induced by urbanization (e.g., soil sealing, canalization of creeks), the true significance of urban land use becomes obvious when considering urban-rural linkages (Lambin et al. 2001). Cities depend on land-intensive systems for food, energy, and other natural resources intensifying land use change in the surrounding rural environment. This link might become spatially uncoupled in a more-and-more globalized world and can put even higher land use intensification pressure on basins with favorable conditions for agriculture or forestry.

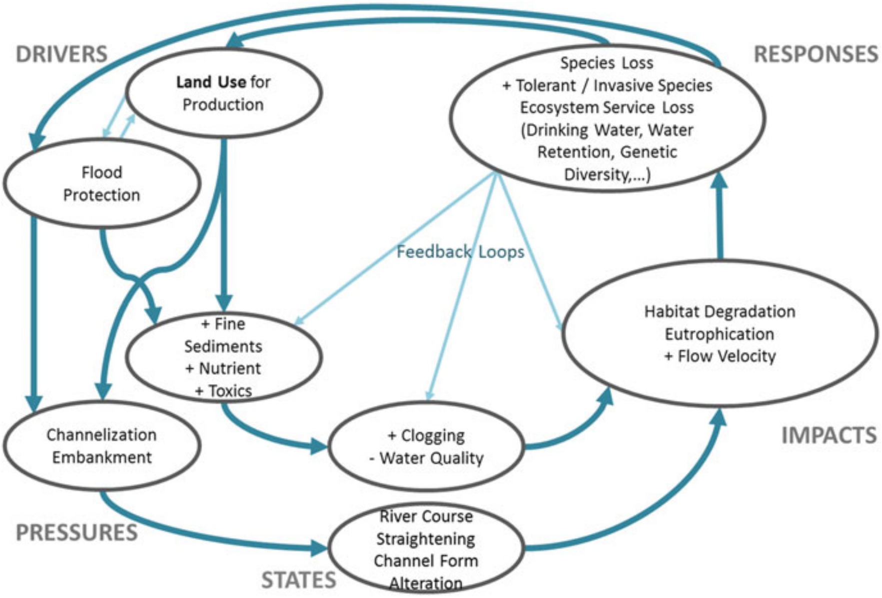

Fig. 13.1 DPSIR framework for land use and human alteration with impact on aquatic ecosystems

In particular, agriculture triggers several processes with negative effects for ecological integrity, not only for river stretches directly adjacent to farmed fields. Fertilizers are deployed over large areas. They are washed out by precipitation and transported into the water column of rivers, raising nutrient levels. This effect accumulates, increasing nutrient concentrations over larger spatial extents. In topographically pronounced, mountainous areas, arable land is situated on the valley floors of rivers. Typically, the floodplains were cut off and drained to make this area accessible for agricultural use. Accordingly, agriculture is often found in the direct vicinity of rivers, reducing riparian vegetation and, hence, the buffering function that reduces surface flows of fine sediments. Combining this lost buffer function with shorter distances between fertilizer and pesticides emissions and recipient aquatic ecosystems downstream increases environmental degradation.

Trautwein et al. (2011) showed that the causal links between land uses and impacts on river fish communities in Austrian rivers are more pronounced than mere correlation implies. The study revealed cumulative effects of several land use categories and quantified the thresholds at which land use's impact on fish communities could be predicted. Agriculture and urbanization were the best predictors for low ecological status based on fish community metrics. Poor and moderate ecological status was observed when >23.3% was classified as agriculture. Urbanization effects could be detected when >2% of the upstream catchment was urbanized. This detrimental effect of urbanization is a disproportionately large effect on instream conditions as compared to other human land uses (agriculture, pasture). Furthermore, effects of agriculture and urbanization were amplified when both categories occurred above their threshold in the catchment.

Multiple impacts from land use are most often coupled to human impacts on the local and reach scale. To develop robust management strategies for riverine ecosystems, the cause-effect chains have to be understood through multiple system components at multiple spatial (and temporal) scales. However, land use characteristics often represent an “umbrella pressure indicator,” i.e., if urbanization or intense agriculture occurs, then also other human-induced pressures can be found. Besides nutrient and pesticide pollution, agriculture is often linked to increased water use, fine sediment input, and habitat loss due to physical modifications (see Fig. 13.1). Agricultural intensification along rivers induces channelization to increase the area of arable land (e.g., a so-called "10th federal state of Austria" was created by draining wet meadows after World War II) and to prevent flooding of the crops. However, channelization itself affects the functioning of the riverine ecosystem in different ways; on the one hand, it degrades the morphology; on the other hand, hydrology is also affected.

The synergistic and antagonistic effects of multiple pressures are still not fully untangled (Schinegger et al. 2012) despite much ongoing research (e.g., Nõges et al. 2016; Piggott et al. 2015; Schinegger et al. 2016). As many catchments face pressures related to more than one land use category (e.g., agricultural and urban land) as well as multiple human pressures, it is essential to understand in detail the different impact pathways and the interrelationships between the different pressures to develop robust management strategies.

(Best) Management Practice for Mitigating Land Use Effects

Managing land use effects can take place at different scales. At local scales (i.e., small spatial extent), it is more feasible to plan and implement land use with less detrimental effects than on the catchment scale. For example, near natural vegetation can be sustained by a variety of extensive land uses in riparian and floodplain area. Riparian zone management was one of the four most applied goals in 37,000 river restoration projects in the USA (Bernhardt et al. 2005).

In fact, riparian zones are critical transition areas between streams and their catchments. Riparian management can effectively influence stream conditions directly and buffer detrimental effects of land use practices further away from the river. The benefits promise to be highly disproportionate to the land area required, especially at the local and reach scale. For example, the protective effect of riparian forests against mixed agricultural and urban pressures was demonstrated in three regions in France. Riparian corridors appear to be manageable areas, and these results strongly support the idea of including their restoration in priority actions for achieving good ecological status (Wasson et al. 2010). However, in a nested hierarchy of landscape types, large-scale uses can constrain and even overwhelm smaller-scale processes. For example, the cumulative impacts of intensive land use over entire catchments are likely to override riparian zone mitigation (Hughes et al. 2006; Wang et al. 2006).

Land use management across entire catchments is much more challenging. Reversal of land use to a less-developed state over vast areas is usually economically and politically infeasible. However, mitigation of land use effects can be accomplished by promoting best management practices (BMPs) and improvements in landscape management. Modernization of agriculture can reduce and change fertilizer applications in the catchment and reduce soil erosion through conservation tillage and cover crops. Mitigation measures in urban areas address stormwater runoff and input of toxics from point sources.

One example of runoff management in small rural catchments is provided by Wilkinson et al. (2014). They tested runoff attenuation features (RAFs) based on the concept of the storage, slowing, filtering, and infiltration of runoff already at its source. RAFs are various constructions like bunds, drain barriers, runoff storage features within the main channel, and adjacent and large woody debris dams. By placing multiple measures all over the catchment, both flood risk and fine sediment input to streams could be reduced and improvements for water quality and stream ecological status could be achieved. While the high performance of RAFs can be achieved at relatively low cost for installation, to effectively manage runoff at larger scales would require a distributed network of RAFs, and the transactional overhead of negotiating with multiple land owners and government agencies can be quite challenging.

The previous example of RAFs well illustrates that restoration actions are expected to be most effective when they follow process-based principles by addressing causes of degradation in line with physical and biological potential at the appropriate scale (Beechie et al. 2008, 2010). Still, they do not solve root causes of the detrimental effects, because surface runoff and erosion from arable or pasture land still occur and the features need maintenance following floods regularly.

13.5 Research Outlook

Much research is needed to identify and test specific mechanisms that link human practices with aquatic ecosystems over multiple scales. This is especially difficult over vast areas, e.g., large river catchments. Well-defined hypotheses of cause-and-effect networks have to be formulated for testing models with empirical data and improve predictions. The lack of such mechanistic hypotheses was already outlined by Johnson and Gage (1997) and reviewed again by Allan (2004) as the majority of studies report correlative relationships between landscape and instream conditions but do not permit specific predictions of instream responses. Current knowledge is therefore limited to apply prescriptive management.

Two areas should be exploited to significantly advance the research frontier. First, we should exploit the increasingly available remotely sensed data to develop synoptic evidence at multiple scales, especially macro-scales. Large-scale, long-term data of multiple investigation sites should be assembled in large (global) datasets and linked to global environmental observation data that are produced in increasing resolution and rate at relatively less cost compared to field investigations.

Second, such macro-data should be used to challenge a new generation of hypotheses that are better formulated to explore causal mechanisms at larger scales and across scales, e.g., micro up to macro and vice versa. This should help uncover mechanisms about how streams interact with their surrounding landscapes. Better hypotheses will grow out of improved theoretical ecological models of dynamic landscapes and species distribution and evolution, developed to predict ecosystem dynamics. With this move forward, also management actions will better ensure rehabilitation success. Nevertheless, the scientific challenge will persist, because more complex models with increased number of factors are more difficult to interpret. Solving models with higher complexity is achieved on higher computational cost and at expense of tractability.

Large-scale, long-term experiments would be ideal to test mechanistic relationships between land and stream but they are costly and extremely difficult to design and manage. Instead, a promising way to go is still via analyzing "natural" experiments (i.e., space for time substitution, natural disturbance comparison, before and after policy change) (Paulsen et al. 2008).

From the management perspective, it is of considerable interest to distinguish the effects of various pressures within the catchment. Very often multiple pressures occur simultaneously, but to identify which one is most important is a difficult task. Von der Ohe and Goedkoop (2013) could distinguish the effects of co-occurring pressures on benthic invertebrates by stressor-specific metrics. But when effects of pressure interaction are not additive but antagonistic or synergistic, the untangling and identification of such specific metrics become very challenging (Schinegger et al. 2016). Development of ecological assessment methods that are able to indicate most important pressures and even pressure combinations is still highly desired by water managers (Hering et al. 2014).

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