INCORPORATING DIFFUSE POLLUTION ABATEMENT INTO WATERSHED MANAGEMENT - WATERSHED VULNERABILITY

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ABSTRACT
Restoring and maintaining the water body integrity is the goal of water quality and diffuse pollution control efforts both in North America (TMDL process) and Europe (Water Framework Directive). The concepts of water body and watershed (land) integrity are outlined. Diffuse pollution is caused both by pollutant discharges from land sources as well as land use and other activities that impair habitat, modify hydrology or adversely impact riparian zones. Current approaches to watershed management plans require establishing the ecologic potential of the water body which is closely related to the condition and vulnerability of the watershed. On a regional (watershed) scale, diffuse pollution is a result of the conflict between the watershed retention capacity for pollutants and input or emissions of the pollutants within the watershed. The watershed retention capacity is related to the capacity controlling parameters. The watershed vulnerability dassification identifies watersheds producing or being susceptible to producing excessive diffuse pollution. Watershed vulnerability is affected by the various watershed morphological, land use, hydrological characteristics and land use characteristics. A hierarchical model can link the biotic integrity endpoints to the habitat, water and sediment risks, and to the stressor causing the risks. Developing watershed vulnerability classification and models linking the biotic integrity end point with watershed stressors will enhance targeting and prioritization of watersheds requiring abatement of diffuse pollution.

Key words: Integrity of receiving waters, Ecologic potential, Total Maximum Daily Load, Water Framework Directive, Watershed vulnerability, Watershed Classification, Water quality risk, Habitat risk, Watershed planning, Watershed modeling

IMPACT OF DIFFUSE POLLUTION ON INTEGRITY OF WATERS
There are multiple root causes of damages to the ecological status of surface and groundwater resources (impairment of integrity) and their diminished uses for humans (Figure 1). The point and nonpoint loads of pollutants from the watershed or direct point source discharges are one source, habitat degradation by stream modification and change of lands surrounding the water body are another cause. These stressors create a risk or a probability that aquatic species indigenous to the water body will disappear. At the same time, the stressor may cause increased risk to public health due to people eating contaminated fish, drinking contaminated water and due to the risk of gastrointestinal disease to those who want to use the water body for swimming and other contact recreation. The ultimate result is the degradation of the aquatic ecological system exhibited as the disappearance of species of organisms that would otherwise thrive in the unimpacted water body and a loss or impairment of the beneficial uses of the water body for humans.

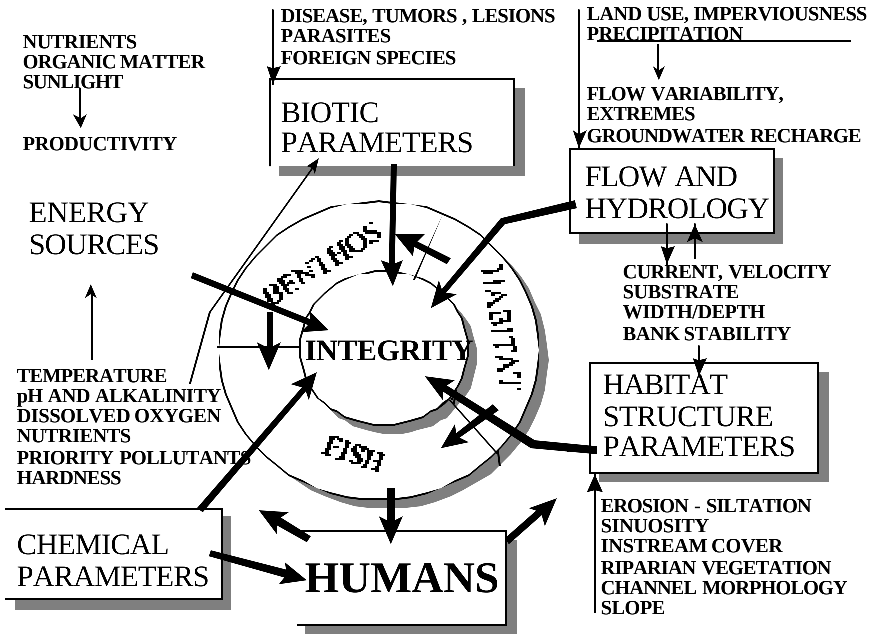

Figure 1 Concept of integrity of receiving waters (adapted from Karr et al., 1986)

The good ecological status of the water body, called “integrity,” has been defined as the ability of the water body ecological system to support and maintain “a balanced integrated, adaptive community or organisms having a species composition, diversity and functional organisms comparable to that of natural biota of the region" (Karr et al., 1986). However, in many areas, human activities have radically altered the landscape and the aquatic ecosystem, such that an attainment of the predisturbance ecologic conditions of the watershed and the water body is impossible (Committee, 2001). Establishing the ecological potential of the water body, while considering irreversible and reversible changes in the watershed, is the goal of the European Water Framework Directive (WFD) and also of the US watershed management programs required by the Clean Water Act (CWA).

"Integrity" of a water body has three dimensions, physical (habitat), chemical (water and sediment composition), and biological. Indices of Biotic Integrity (IBIs) for assessing this three dimensional integrity have been developed and implemented (Barbour et al., 1997). In the US, both fish diversity and numbers, macroinvertebrate and habitat assessment indices and criteria are used in addition to chemical assessment and criteria/ standards. A macro-invertebrate index is similar to that used in Europe where it has almost a 100 year tradition (Kolkwitz and Marson, 1908).

In the context of preserving and restoring the integrity of the receiving water bodies, being the goal of CWA and WFD watershed programs, the term pollution has a broader meaning that was already embedded into the CWA. This definition defines pollution as impairment of integrity caused by humans. In a broader meaning of this term "pollution", as defined in the CWA, includes excessive loads of pollutants from point diffuse sources (sediment, nutrients, biodegradable organics, toxics, heat), physical adverse alteration of the water body integrity such as channel lining and straightening and impoundment, cutting down trees lining the water body, loss of riparian habitat, drainage of riparian wetlands, or hydrologic modifications in the watershed that increase flow or temperature magnitudes and variability. The latter cases that do not include discharges of pollutants, if they are wide spread, could be considered as diffuse pollution.

WATERSHED VULNERABILITY

Waterbody vulnerability is essentially a conflict between the atmospheric and terrestrial (point and nonpoint) loads to the watershed, watershed retention capacity (WRC) and loading capacity (LC) of the water body that cause the water body integrity to be threatened or impaired. This is the foundation of the Total Maximum Daily Load (TMDL) program in the US. Because in most cases point sources directly discharge into the receiving water bodies, watershed vulnerability then implies a situation wherein waterbody integrity is threatened mostly by diffuse pollution originating from the watershed. As stated above, water body modifications affecting habitat also impact the biotic composition of the water body and must be considered as a stressor (pollution).

Borrowing the relationships of the TMDL concept, the conflict arises when the load exceeds the loading capacity, or

$$\text{Vulnerability index} = WVI = WL + NPL + NL - WRC - LC \tag{1}$$

where WVI = watershed vulnerabilty index defined herein for illustration; WL = point source waste load; NPL = nonpoint source load; NL = natural load; and LC=loading capacity of the water body. The background load, being of anthropogenic origin, is included in the nonpoint load NPL. In the TMDL logic, if WVI is positive or could become positive in the near future, the water body is classified as water quality limited (USEPA, 1991; Committee, 2001).

This is obviously an oversimplification that will be addressed by research. In the concept of the broad watershed classification the effect of both nonpoint and background loads can be related to the watershed retention capacity (WRC) that is different from the water body loading capacity. Watersheds can retain majority of pollutants in soils and wetlands. Often more than 99% of incoming potential pollutants reaching the watershed from atmospheric and terrestrial sources, local and distant, are safely retained by soils and other media (Salomons and Stol, 1995) and only a small portion moves towards the receiving waters by erosion, ground-water seepage, washoff, etc. This excess then constitutes diffuse pollution originating from the impervious parts of the watershed (urban and transportation), from agriculture, and other modified lands. However, even this small fraction can be detrimental to the aquatic biota. As a matter of fact, diffuse pollution is responsible for more than 50% of noncompliance with the water quality goals specified by the Clean Water Act.

WRC of pollutants is affected by the same factors as watershed hydrology. This concept was introduced in Europe (Salomons and Stol, 1995). However; unlike the hydrologic factors that without a change of human interference are invariant, the WRC has a limit that is not invariant. The concept is show on Figure 2. The watershed retention capacity for pollutants is related to the Capacity Controlling Parameters (CCPs) that include organic matter content of soil and surface vegetation, Cation Exchange Capacity, Redox Potential, soil adsorption capacity, and others. Shallow bedrock geology and soil texture are also important in determining the CCPs. Generally sandy soils have smaller retention capacity than finer soils in the silt and silt loam category. Calcareous finer soils overlying carbonate sedimentary bedrocks have a better buffering capacity than non-calcareous soils overlying crystalline or sedimentary non-carbonate consolidated or non-consolidated geological layers. Adsorption and the retention capacity of soils with high clay content are limited by permeability. In general, soil pH, clay content, and organic content are good surrogates to which the CCP can be correlated. Soils rich with organic matter have a very high retention capacity for organics and many pesticides but relatively low retention of phosphorus and nitrogen (especially in nitrate form) and mobile toxic pollutants. The presence of wetlands increases the WRC.

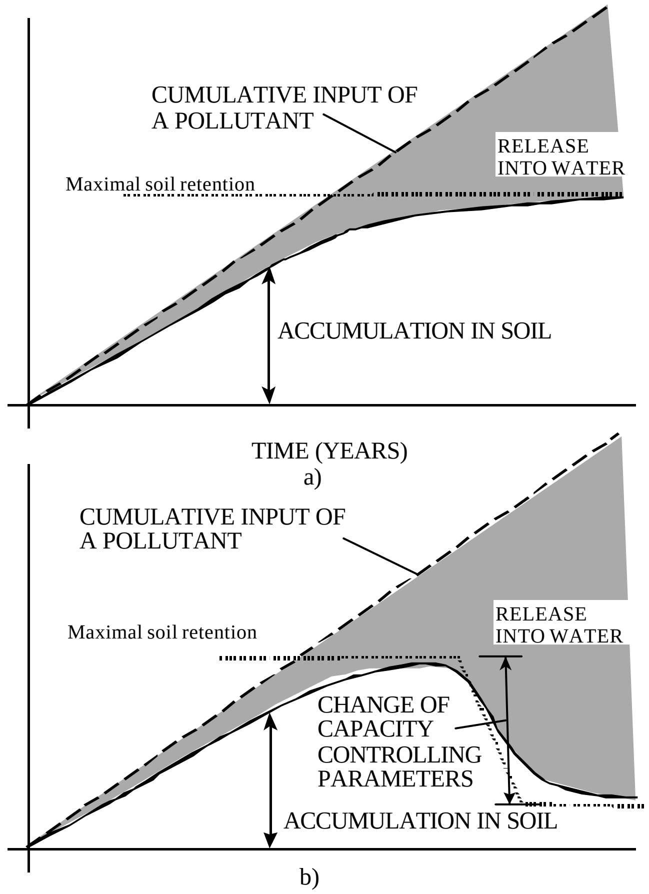

Figure 2 Concept of capacity controlling parameters in a watershed, (a) constant storage maximum, (b) storage affected by variable CCPs (Novotny, 2003)

The collapse of the watershed retention system leading to a large increase of pollutant loads from the watershed can occur when (a) the steady large load of nonpoint pollutants inputs exceeds the retention capacity of the watershed, and/or (b) as the result of nonpoint inputs and/or climatic changing the CCP factors change (Salomons and Stol, 1995; Novotny, 2003; Stigliani and Salomons, 1993).

There are locations where such changes of CCPs have occurred and resulted in increased loads of pollutants that exceeded the loading capacity of the receiving water bodies. In some cases, such changes could be characterized as a collapse of the WRC. The most notable are changes caused by regional inputs of acid rainfall in Central Europe that resulted in massive deforestation of watersheds with subsequent loss of soil and pollutants from the soil. Also research in the New England Coastal Basins has revealed (Flanagan et al., 1998) that streams in the mountainous watersheds have very low alkalinity (Figure 3) and low pH and are generally soft, which may increase the toxicity of metals. These streams and watersheds are affected by acid precipitation. For example, hydrogen ion and sulphate deposition to the Hubbard Brook Experimental Forest (New Hampshire White Mountains) are three times larger than the watershed can safely assimilate (Likens and Borman, 1995), resulting in low pH and subsequent adverse effects on fish density and composition. Northeastern pristine watersheds have very low WRC for pollutants and acidity in the snowmelt. The first flush of pollutants from snowmelt, including hydrogen ions, can be devastating to the ecology of such watersheds. Elevated acidity of rainfall also increases elutriation of metals from soils and the urban infrastructure (Novotny, 2003). Consequently, fish assemblages and aquatic fauna even in pristine New England streams are relatively poor and the streams do not provide conditions for spawning (e.g., atlantic salmon).

Reduction of WRC also occurs when wetlands are drained or native undisturbed lands are developed. For example, change of prairies into arable lands and drainage of wetlands triggered nitrification of large amounts or organic nitrogen stored in the soils and released large amounts of nitrate into groundwater and subsequently into surface water bodies. Kreitler and Jones (1975) documented that in Texas watersheds where such land changes had occurred, the nitrate content of groundwater and recipient streams increased up to 250 mg/L. Nitrification in soils is a major contributor to the nitrogen content of the Mississippi River, causing hypoxia in the Gulf of Mexico (Burkart and James, 1999). On the other hand, conversion of agricultural lands into forests in Holland (a cornerstone of the USDA conservation programs) changed the pH of the soil and released large elevated quantities of metals (Salomons, 1995). Urbanization that removes the soil adsorption from the WRC is also a dramatic adverse change that results typically in orders of magnitude increases of pollutant loads from the watershed.

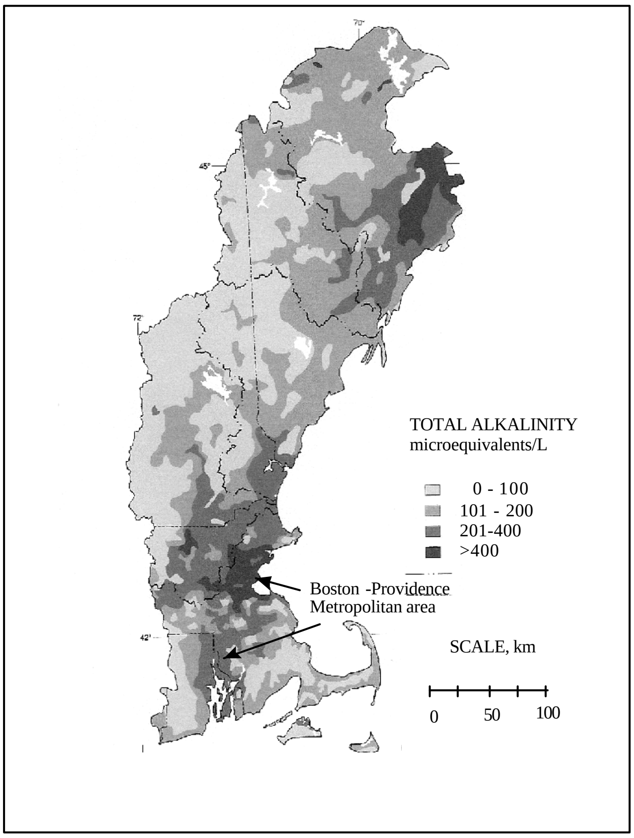

Figure 3 Alkalinity of the streams in the Northeast US Coastal Basin. From USGS

In general, the large scale processes that effect the WRC and, therefore, the diffuse pollution loads are:

- Change of soil acidity and loss of buffering caused by acid precipitation and deposition (wet deposition represents only a portion of the acid input)
- Deforestation
- Land conversion (e.g., to agriculture or urban use)
- Wetland drainage
- Change of soil buffering and adsorption capacity due to effects other than acid inputs
  - Loss of soil organic matter due to agricultural practices and urban construction activities
  - Loss of top soil by wind and water erosion
  - Change of redox conditions by drainage
  - Increasing imperviousness of the watershed by urbanization
- Surface extraction of minerals and coal

Thus, the change of CCPs and the storage capacity of the watershed for the pollutants can be triggered by land use change, atmospheric and nonpoint pollution inputs to land and riparian corridors. Land use changes affect hydrology, WRC and nonpoint loads (for details see Novotny, 2003). The parameters affecting CCPs and WRC can be obtained from the nationwide and regional standard data bases and often are available in GIS format. A functional relationship for the WRC based on the key soil and land use parameters can be obtained from the literature and/or developed. For example, a functional relationship for soil adsorption maximum for phosphate related to soil pH, organic carbon and clay content is included in Novotny (2003) and its development is described in Novotny and Chesters(1981). Retention capacity for metals and nitrogen can be related to the degree of saturation of soils (wetlands vs. dry lands) and soil pH, and pesticide retention will be related to the organic carbon content and biodegradability (half life) of the pesticide. CCPs can also be related to soil texture and depth of the soil to the bedrock and type of the bedrock. Unlike the previously developed *constant export coefficients*, (e.g., Reckhow et al., 1980; Beulack and Reckhow, 1982; Novotny and Chesters, 1981) the diffuse pollution loading coefficients should and will be linked to the difference between the variable WRC related to watershed land use, geology and CCPs and the cumulative load (deposition) of pollutants to the watershed.

There is evidence that the watershed modification impact is less for streams that have good riparian buffers (Flanagan et al., 1998). Also a change of watershed from agriculture to urban has less impact (i.e., the watershed is already degraded) than change from forest to urban. Thus, the risk due to land use changes has to be stratified and scaled. (Novotny, 2003).

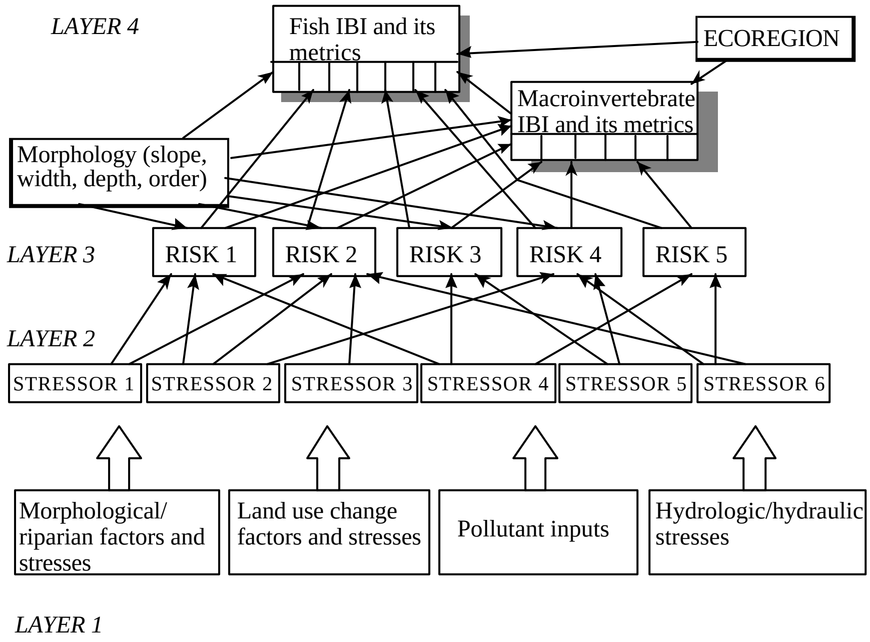

Figure 4 Hierarchical model and watershed classification schematics

Flow changes. The land use changes also have an impact on the hydrology of the watersheds. Flows that in the predevelopment period had a recurrence interval of 100 years may become 2 year flows when the watershed is fully developed. Urbanization makes flows greater than a bankfull more frequent, resulting in unstable and enlarging channels. Similarly, land use changes also diminish the base flow. The result of urbanization is enlargement of the channel with less or no base flow. Profound hydrologic changes can also result from deforestation, land conversion to agriculture, agriculture to urban and transportation.

LINKING DIFFUSE POLLUTION TO IMPAIRMENT OF INTEGRITY

Each Index of Biotic Integrity (physical, benthic and fish) is a composite of metrics that provide a numeric input value for the index . Ecologically, the fish index has the highest trophic value of the three biotic indices. The physical (habitat) index represents a risk of habitat impairment and should be considered a layer below the two biotic indices. If properly designed, the benthic invertebrate index of the biotic integrity should reflect the quality of the upper sediment layer and the chemical fluxes through the upper sediment/water interstitial layers. The fish IBI reflects the longer term water quality but it could also be impacted by the quality and numbers of benthic invertebrates that are food to some fish species, by the quality of the habitat, by the variability of the waters quality and frequency of short duration chemical shocks.

The problem with the use of biotic indices is the fact that they reflect and represent the effect of many stressors (Figure 4), characterized under the expanded definition of diffuse pollution, and the separation of the stressors' effect is very difficult. Since the early developments of the biotic indices classifying the status of receiving waters, researchers attempted to link the indices to various abiotic or morphological attributes and parameters of the receiving water body (e.g., Karr and Chu, 1999, Wang et al., 2000). The most frequently used relationship published in literature was the relation of the fish or macroinvertebrate Index of Biotic Integrity (IBI) to percent urbanization or percent imperviousness (e.g., Karr and Chu, 1999) or other land uses within the watershed (Wang et al., 2000). Linking biotic integrity indices and their matrices to single morphological variables (e.g., imperviousness or population density) is simplistic and does not describe the cause-effect relationships. Nevertheless, it is useful for illustrating the problem. A layered, hierarchical model and analysis are necessary to quantitatively and qualitatively identify the linkages between the stresses and biotic endpoints. Finding linkage between multiple stressors and biotic endpoints is now in the research phase sponsored by the US EPA/NSF/USDA STAR (Science To Achieve Results) research program. An interdisciplinary research at Northeastern University and other research centers is addressing this challenge. The model being developed by the Northeastern University research is based on the principle of risk propagation from the stressors causing pollution (according to the broad, inclusive definition), pollution, risks and the effects of risks on the probability that species will be lost from the aquatic ecosystem that will impact the integrity. The model has four layers of interactions (probabilistic risk propagation) as shown on Figure 4 and will use artificial neural network to derive the linkages and relationships.

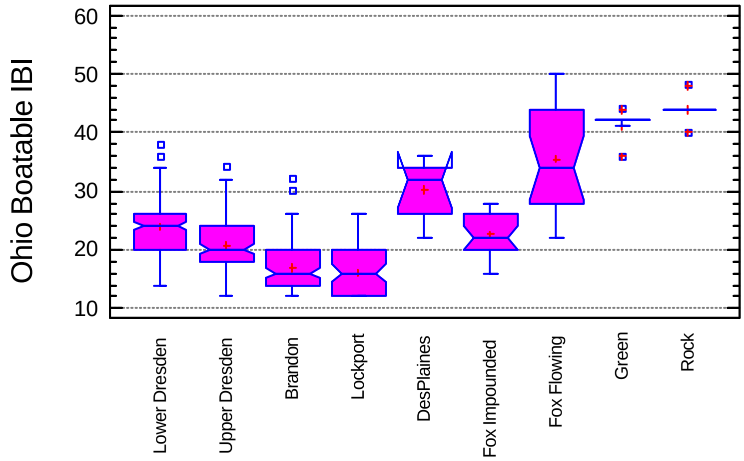

Figure 5 Comparison of IBIs for impounded and free flowing streams in Northern Illinois. Data provided by the Illinois Environmental Protection Agency, US Environmental Protection Agency (Region V) and EA Environmental Systems. Plot generated by T. Ehlinger

Effect of stream modification (e.g., impoundments). The impact of human changes to channel configurations must also be considered. Changes such as straightening or lining channels often effectively destroy the habitat, but even less dramatic changes such as impoundments can have a significant impact. Figure 5 presents the fish IBI data for Northeastern Illinois. Brandon and Lockport pools are highly modified navigation channels of the Des Plaines River which is a part of the Upper Illinois River waterway. The effluent dominated river receives most of the flow and effluent discharges from the metropolitan Chicago area. The upper Brandon pool has a highly impaired to a non-existent habitat, sediment contamination and low dissolved oxygen. The Lower Dresden pool of the Des Plaines River has better habitat conditions but suffers from thermal pollution. The Lower Des Plaines River does not meet the ecological potential of impounded and modified streams exemplified by the reference Rock and Green Rivers; however, these reference streams have IBI scores that are still below those of wadeable reference streams (IBI of 55 to 60). Of note is the USEPA unpublished study on another impounded river, the Fox River, wherein concomitant IBI measurements were made 0.5 km downstream (free flowing) and 0.5 km upstream (impounded) of low head dams. The IBI data from the USEPA Fox River study shows the significant impact of low head dams on the biotic integrity. Impounded river reaches have significantly lower IBI scores than free flowing reaches. Impoundments without fish passage also fragment the habitat and prevent migration of fish

Hydrologic and morphologic deviations from reference conditions are also caused by water withdrawals, wastewater discharges, hydropower production (peaking), and operation of navigation locks. For example, fast flow variations of 30 - 60 m^3/sec were observed during the operation of locks on the Lower Des Plaines River in Illinois. Similar rapid flow variations were observed in the modified stream of the New England Coastal watershed (peak power production).

CONCLUSIONS

The majority of adverse changes in watersheds impairing water quality occurred over a period of thirty to more than one hundred years. For example, in the second half of the 19^{th} century, a major part of the watershed of the Lagoon of Venice in Italy or large portions of the states of Illinois, Indiana and southern Wisconsin consisted of wetlands. In a period of about 80 years most of the wetlands were drained, cultivated and converted to agricultural and urban uses. This conversion changed the redox status of the soils and soil cover with a concurrent large increase of suspended solids, dissolved organic matter, nitrogen, and phosphate loads from the watersheds located in the affected regions. Changes due to the acid rainfall in Central Europe, Scandinavia or northeastern US have been documented over the last fifty years. The dramatic nitrate increases in European streams had reached alarming levels over a period of thirty years in the second half of the twentieth century. Some changes are sudden. For example, dramatic reductions of fertilizer and pesticide inputs occurred in Eastern Europe over a period of less than three years after the regime changes in 1989-1990.

Watershed vulnerability is a new concept of analyzing causes and impacts of diffuse pollution. Watershed vulnerability is affected by the capacity controlling factors and other morphological, hydrological, and ecoregional characteristics of the watershed. Developing watershed vulnerability classification will enhance targeting and prioritization of watersheds requiring abatement of diffuse pollution. The watershed vulnerability classification identifies watersheds producing or being susceptible to producing excessive diffuse pollution (caused by pollutants and other types of pollution) that would result in unacceptable impairment of the ecological integrity of the receiving water bodies. The hierarchical risk propagation models may identify the major stressors and enables selection of the best management practice aimed at reduction of the causative risks and stressors. The watershed wide pollution control efforts in North America (TMDL) and

Europe (Water Framework Directive) envision targeting watersheds and bringing the receiving water bodies up to their ecological potential, i.e., restoring and maintaining the integrity of the receiving water bodies.

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