Chapter 9: Impacts of exurban development on water quality

Authors: Kathleen A. Lohse and Adina M. Merenlender

"Nothing is more fundamental to life than water. Not only is water a basic need, but adequate, safe water underpins the nation's health, economy, security, and ecology. The strategic challenge for the future is to ensure adequate quantity and quality of water to meet human and ecological needs in the face of growing competition among domestic, industrial-commercial, agricultural, and environmental uses" (National Research Council 2004)

INTRODUCTION

Sustaining water resources has emerged as one of humanity's greatest challenges (National Research Council 2004); it requires managing existing threats to surface and ground water quantity and quality as well as planning for future impacts due to changes in land use and other global changes (Butcher 1999; Fitzhugh and Richter 2004). In many cases, however, resource managers, conservationists, and planners have a limited understanding of the impacts of human activities on stream and ground water conditions. In particular, little is known about how different types of urban land use, especially low density housing development or exurban growth, affect stream and groundwater conditions because tools to detect and quantify exurban development, such as commonly used remotely sensed imagery (e.g., Landsat), cannot adequately characterize differences in housing densities. Consequently, exurban development is often excluded from model development and risk assessment. In addition, there is often large uncertainty in how future type and extent of land use will impact these conditions. Together, these factors make it challenging to predict how watersheds will respond to future changes and other unforeseen interactions (Nilsson et al. 2003). In this chapter, we contend that despite these large uncertainties, coupled land use impact--land-use change models can be important decision support tools to identify areas that are more sensitive to land-use change, to examine the inherent trade-offs associated with various policy options, and to identify options that result in water and land conservation.

Agricultural and urban land use activities are often considered key drivers leading to water quality impairment of streams and other water bodies (US EPA 2000). However, exurban growth is increasingly recognized as an emerging development pattern in critical need of ecological and water quality assessment (Theobald 2001; Theobald 2004). Indeed, exurban development is the fastest-growing land-use type in the United States (US). (Heimlich and Anderson 2001; Theobald 2003; Brown et al. 2005) and is expanding in Canada and Europe (Dubost 1998; Azimer and Stone 2003). For example, recent nighttime aerial analyses of the conterminous US have revealed that exurban development covers 14.3% of the US and represents 37% of the population; urban areas make up 1.3% and represents 54.7% (Sutton et al. 2006). More importantly, urban and exurban development represent fundamental different types of growth (Newburn and Berck 2006). Whereas urban development requires sewer and water infrastructure before higher-density development (<1 acre/house) can be built, exurban development (5-40 acres per house) is almost invariably serviced by private wells and septic systems and, thus, not bound to existing or planned sewer and water service areas (SWSA). These differences between urban and rural-residential development extend the possible range and associated environmental impacts of rural-residential development such as sedimentation but also temperature, organic wastewater contaminants and nutrient loading from septic systems well beyond the urban fringe (Hansen et al. 2005; Newburn and Berck 2006; Lohse et al. 2008). Together, differences in factors controlling urban and exurban development and their differential land-use impacts indicate that planners and watershed managers will need to determine the relative effects of exurban versus urban development. Land-use change models will also need to distinguish between these different residential densities to forecast land-use development patterns.

Coupling land-use impact models and land-use change models is emerging as a powerful decision support tool for watershed manager and planners to help inform stakeholders about inherent trade-offs associated with various land use and policy decisions as well as assist in guiding conservation planning and development. Different scenarios can be generated from these integrated models, and scenario planning can assist with environmental problem solving when there is a high level of uncertainty about the system and manipulating the system is difficult (Peterson et al. 2003). The difficulties inherent in integrating land-use impact models used by hydrologists, geomorphologists, and ecologists, and models of land-use change used by planners and economists are described by Nilsson et al. (2003). In brief, Nilsson et al. (2003) list a number of improvements that need to be tackled by each discipline in order to develop useful integrated models; until then, they argue that these disciplinary limitations will prevent the development of useful integrated models for meaningful forecast relationships between future expected land-use change and stream ecosystem responses. However, we believe that language and epistemological barriers are rapidly being lowered by scientists who are increasingly cross-trained in interdisciplinary environmental sciences. This new breed of scientists has the tools and language to resolve many of the issues that have plagued earlier attempts at integrated modeling.

In particular, spatially explicit economic modeling is rapidly evolving to improve simulation models of land-use change and consequent development patterns. Also, advances in land-cover mapping via object recognition and high-resolution imagery, as well as a commitment on the part of local governments to map cadastral data, have led to rapid improvements in source data for land use and environmental modeling. So, while we acknowledge the need for continued advances in each field to improve our disciplinary understanding, integrated models can be developed that are useful to inform decision makers about inherent trade-offs associated with various policy decisions and to identify future land-use and water management scenarios that are more sustainable. With these limitations and goals in mind, we must follow the example set by climate modelers and improve our modeled predictions, while explicitly addressing uncertainty in resulting scenarios so as not to weaken our credibility with inaccurate predictions.

In this chapter, we present our current understanding of the impacts of different urban land use types on water quality. We then outline how interactive decision-support tools or models can be developed to minimize current and future land use impacts on water resources. Essential steps in this model development process include: (1) quantifying relationships between land use and water quality/quantity (land-use impacts), (2) developing land-use change scenarios that forecast likely future land-use change scenarios, 3) developing economic valuation models, and 4) integrating these models (land-use impacts, land-use change, and economic valuation) to evaluate environmental and economic trade-offs (Figure 9.1). We use a case study to demonstrate the utility of this approach to guide land-use planning that will ultimately improve water quality and water security for human and natural systems. Our approach emphasizes that parcel-level data can be used as the fundamental unit of land-use change to integrate and build forecast models, resolve geographic resolution issues between models, and permit ecologists to detect relative effects of low, medium and high-density housing as well as other land uses on watershed processes.

REVIEW OF URBAN AND EXURBAN LAND-USE IMPACTS ON WATER QUALITY

Here we provide a brief review of the scientific literature focused on the impacts of urbanization on water-quality characteristics including nutrients, organic pollutants, metals and sedimentation. Conversion of land to exurban and urban housing development also alters hydrology and these impacts are detailed in Chapter 11 of this volume. We focus this review on differences in housing densities from exurban development (5-40 acres per house) to urban (<1 acres per house). In this chapter, we do not address a recent and growing trend in new community developments in which developers build towns consisting of commercial, retail and residential land uses outside of existing city boundaries because the development density is similar to urban areas, and we expect the impacts of these new town developments will be similar to urban development.

Water Quality Assessments and Sources

Assessments. States, U.S. territories, and other jurisdictions are required by Section 305b of the Clean Water Act to assess the quality of the surface and groundwater and report their findings to the US Environmental Protection Agency (EPA). Water bodies are then classified according to water quality standards and their ability to meet designated uses, including aquatic life, fish consumption, primary contact recreation, secondary contact recreation, drinking water supply, and agricultural use. For example, surface or groundwater quality must not exceed the specified Maximum Contaminant Level (MCL) established by the US EPA for each of the regulated water quality constituents to meet drinking water standards. Similarly, surveys of sediments are performed to estimate the probability of adverse effects of contaminated bed sediments on aquatic and human life as required by the Water Resource Development Act of 1992. The contaminant concentration in bed sediment that is expected to adversely impact benthic (or bottom-dwelling) organisms has been specified as the Probable Effect Concentration (PEC).

Based on these standards, the US EPA reported that 35,000 miles of river were impaired by urban runoff and sewers in the US in the year 2000. An additional 28,000 miles were impaired by municipal point sources, and 129,000 miles of river were impacted by agricultural land use. Figure 9.2 shows that the leading source of impairment of rivers was pathogens (bacteria) followed by siltation, nutrients and metals. Figure 9.2 also shows that of the surveyed sediment (21,000 sites), 26% of the sediment samples were classified as Tier 1, indicating that the observed contaminant concentrations were likely to have adverse effects on aquatic life and possibly human health. Another 49% of the sites were classified as Tier 2 indicating possible, but infrequently expected, adverse effects on aquatic ecosystems or human health. The most frequently observed contaminants in Tier 1 were toxic organic compounds, specifically polychlorinated biphenyls (PCBs) (20%), followed by pesticides, mercury (Hg) and then another class of toxic organic compounds, polycyclic aromatic hydrocarbons (PAHs). Metals were the most frequently encountered contaminant in Tier 2 (58%).

Sources. Sources of contaminants can be differentiated into point and non-point sources. Point sources refer to discharge of contaminants from a specific location such as municipal discharges from waste water treatment plants. Non-point sources are contaminants from more diffuse sources and can be differentiated into atmospheric and fluvial sources. Atmospheric deposition is commonly associated with organic pollutants such as PCBs and PAHs and metals such as lead (Pb) and mercury (Hg) derived from coal combustion, burning of leaded gasoline and other petroleum products. Non-point sources from watershed-derived fluvial transport include runoff from agricultural fields, mine drainage, or urban areas such as pavement, lawns and golf courses, and leaking septic tanks. In Table 9.1, we highlight sources of a select set of contaminants and the predicted enrichment factor from conversion from rural to suburban to urban land use. In the next sections, we examine in more detail the impacts of urbanization on occurrence of different contaminants including organic compounds such as biological pathogens and toxic organic compounds, inorganic compounds including nutrients and metals, and sediments. We suggest best management practices to watershed managers, planners and conservationists to minimize sources of contaminants in residential housing developments.

Organic contaminants

Organic contaminants include biological pathogens and other toxic organic substances. Pathogens consist of a diverse group of bacteria, viruses, protozoa, and parasitic worms that are responsible for many water-borne diseases such as gastroenteritis, malaria, river blindness cholera, and typhoid fever (World Health Organization 2008). Although water-borne pathogen associated fatalities remain low in the US owing to safe drinking water and sanitation practices, water-borne diarrhoeal diseases (including cholera) claim 1.8 million lives each year, and malaria claims 1.3 million lives each year in developing counties (World Health Organization 2008). Toxic organic substances include a plethora of human-derived compounds that vary in weight, toxicity, and persistence in the environment (Miller and Miller 2007). Two common classes of toxic organic groups known to pose a threat to human and ecosystem health include polycyclic aromatic hydrocarbons (PAHs) and halogenated hydrocarbons (PCB), and these will discussed in more detail below.

Biological pathogens. Biological pathogens, such as bacteria, protozoa, and viruses, have emerged as primary stressors in surface waters (US EPA 2000) and have recently been found in ground waters (Embrey and Runkle 2006). As mentioned above, the 2000 US EPA report identified pathogens (bacteria) as the leading cause of degradation of rivers impairing approximately 35% of the rivers assessed or 91,431 river miles (Figure 9.2). In a national survey of ground water aquifers, Embrey and Runkle (2006) also showed high occurrence of coliform bacteria; coliform were detected in 33% of the wells sampled. Rather than depth to well, hydrogeologic characteristics and proximity to contaminated sources such as wastewater treatment plants appeared to be more important predictors of occurrence of pathogens in groundwater. These findings have raised awareness of the vulnerability of ground waters to pathogens and the need to understand controls on transport of bacteria as well as viruses to ground waters.

The impact of urbanization on pathogen sources, transport and fate is an area of active research, and researchers are using different tools and techniques to address these questions (see Field and Samadpour 2007 for a detailed review). *Escherichia coli* (*E. coli)* has been increasingly used in monitoring studies as a reliable indicator of fecal contamination (Doyle and Erickson 2006). However, the use of *E. coli* alone as an indicator organism has been questioned because pathogens have been isolated from ecosystems where only low concentrations of fecal coliforms exist (AWWA 1999; Field and Samadpour 2007). More studies are expanding to use molecular techniques to develop microbial source tracking tools to identify the sources (e.g., human, domestic animal, wildlife, and/or bovine) and the pathogenic nature of the bacteria (Field and Samadpour 2007).

To the knowledge of the authors, few studies have been published examining the impacts of urbanization on the occurrence of fecal coliform. One of the few studies on this topic was conducted in Georgia, US and showed that fecal coliform concentrations were significantly higher during base and storm flow in urban watersheds compared to nonurban watersheds. In these systems, fecal coliform typically exceeded EPA review criterion of 400 Most Probable Number (MPN)/100 ml (Schoonover and Lockaby 2006). Land use impact models developed from this work suggest fecal coliform will exceed EPA review criterion when development exceeds 10 and 20% impervious surface cover. Additional studies are warranted in other environmental settings to see if these patterns hold across different hydroclimates and to understand the effects of transport versus source processes controlling the delivery of bacteria and viruses to surface waters.

Until advances are made in terms of understanding the sources, transport, and fate of bacteria, watershed managers and planners can follow best management practices to reduce bacteria sources and delivery to streams. These practices include nonstructural and structural methods. Specifically, nonstructural practices for low density residential development include routine septic inspection and pump-outs and management of pet waste as well as manure from domestic animals. For urban areas, management of pet waste and regular inspection of sewer lines for leaks will reduce unexpected releases of feces into rivers and streams. Structural best management practices include buffers, constructed wetlands, sand filters, infiltration trenches, low impact development, and stream fencing. Examples of low impact development includes permeable pavers, retention areas, grass swales, rain gardens, and minimizing impervious areas to increase runoff infiltration, storage, filtering, evaporation, and onsite detention (*public communications*, Low-Impact Development, www.EPA.gov/owow/nps/lid/). Although these practices can help reduce bacteria in surface water, further studies are needed to evaluate the effectiveness of these practices.

Halogenated hydrocarbons. Halogenated hydrocarbons are hydrocarbons that contain one or more atoms of chloride (Cl), bromide (Br) or flouride (F) and are one of the largest and most important groups of toxic organic contaminants. These chemicals have been linked to adverse effects on aquatic and human health including cancer, reproductive problems, and nervous and immune system problems (Miller and Miller 2007). Sources of halogenated hydrocarbons include solvents, cleansers, and degreasers but also pesticides, electrical equipment, and commercial products. Examples of halogenated hydrocarbon solvents are trichloroethylene (TCE) and chloroform. Prominent examples of halogenated hydrocarbon pesticides are Dichloro-Diphenyl-Trichloroethane (DDT) and 2,4-Dichlorophenoxyacetic acid (2, 4-D). Polychlorinated biphenyls (PCBs) are also chlorinated hydrocarbons often used in electrical equipment. Although polychlorinated biphenyls (PCBs) have been banned in the U.S. since the 1970's, high levels of PCBs can still be found in river and lake sediments owing to their continued use in equipment made with PCBs, their slow degradation rate, and persistence in the environment (Miller and Miller 2007). Dichloro-Diphenyl-Trichloroethane (DDTs) were also banned in the 70's and take a long time to break down, tend to accumulate, and bio-magnify in biota. The persistence of these chemicals was highlighted in a recent national survey of surface waters and ground waters (1992-2001) showing banned (DDT, PBC) and newer organochlorine compounds (chlordane, dieldrin) detected in fish and bed sediments in most streams in the US (Gilliom et al. 2007). In addition, at least one type of pesticide was detected in 90% of the surveyed streams and in 50% of the groundwater wells (Gilliom 2007; Gilliom et al. 2007)

The few studies evaluating relationships between increasing urban intensification (i.e. the conversion from rural to suburban to urban) and particle-associated and water-associated halogenated organic compounds have shown that concentrations of halogenated hydrocarbons increase with the percentage of commercial, industrial, and transportation land use (CIT) in a watershed. Along a rural to urban gradient in the Northeast US, for example, Chalmers et al. (2007) showed higher concentrations of halogenated hydrocarbons in sediments in areas with higher percentage of CIT land use. Land use impact models generated from this work suggest conversion from rural to suburban will result in 2 fold increase in DDT and PCBs and suburban conversion to urban will result in a 3-4 fold increase (Figure 9.3) (Chalmers et al. 2007). Despite the high occurrence of chlorinated hydrocarbons in stream sediments, surveys of urban and reference lake sediment cores (using lakes as long-term historical records of deposition of compounds) show that concentrations of DDT and PCBs in sediment cores are declining over time across the US, suggesting that sources of DDT and PCB in the environment are generally decreasing as these chemicals are being phased out of use and eventually degraded (Van Metre et al. 1997; Van Metre et al. 1998; Van Metre and Mahler 2005). Nonetheless, these studies highlight the persistence of halogenated hydrocarbons in the environment; once released in the environment, these compounds persist for decades to centuries.

Occurrence of other halogenated hydrocarbons such as pesticides and herbicides in surface and shallow ground waters is also high in urban to undeveloped areas. In an USGS study, pesticides were detected in 97% of the surface water and 55% of the ground waters sampled in urban areas (Gilliom et al. 2007). Surprisingly, the study also detected pesticides in 65% of surface waters and 29% of ground waters in undeveloped watersheds. In addition, more than half the fish sampled in undeveloped watersheds contained organochlorines indicating deposition and persistence of these compounds in the environment. In another study across six metropolitan areas, Sprague and Nowell (2008) showed that insecticide concentrations increased significantly with increasing urban cover in low flow conditions whereas herbicides increased with increasing urban cover in three out of the cities examined (Sprague and Nowell 2008). Large agricultural influences in the other three cities appeared to explain herbicide patterns in stream flow.

Best management practices for reducing halogenated hydrocarbons in urbanizing areas include nonstructural and structural practices. Examples of structural practices include reducing halogenated hydrocarbon pesticides and insecticides or using non-halogenated hydrocarbon pesticide and insecticides. Inspection of older electrical equipment and phasing out of older and/or leaking equipment that may contain PCBs is also advised. Structural practices include LID methods and others described in the biological pathogen subsection.

Polycyclic aromatic hydrocarbons. In contrast to PCBs and DDTs that appear to be declining over time, polycyclic aromatic hydrocarbons (PAHs) are on the rise (Van Metre and Mahler 2005). Polycyclic aromatic hydrocarbons originate from natural and anthropogenic combustion of organic material, including fossil-fuel burning in automobiles, power plants, and heating facilities, and have been identified as potentially carcinogenic. They can also be found in creosote, roofing tar and asphalt sealant (Mahler et al. 2005). Indeed, coal-tar-emulsion based and asphalt-emulsion based sealcoats have been recently identified as large sources of PAHs. These sealants are used by many homeowners and commercially and applied to parking lots associated with commercial businesses, apartments, condominium complexes, churches, schools, and business parks, and residential driveways. A study by Mahler et al. (2005) found much higher PAHs in runoff from parking lots sealed with coal-tar based sealcoat compared to those sealed with asphalt-based sealant; the average PAH concentrations in particles in runoff from coal-tar sealed parking lots was 3,500 mg/kg or 65 times higher than concentrations in particles from parking lots not seal coated (54 mg/kg). Average concentrations in particles from asphalt-based sealcoat were lower, 620 mg/kg. The concentration of total PAHs likely to adversely affect aquatic organisms, or the PEC, is 22.8 mg/kg (Mahler et al. 2005). Together, these findings suggest new development and associated vehicular traffic and driveway sealants are major source of PAHs to streams.

Because PAHs are produced as byproducts from partial combustion of fossil fuels and other products, several studies have examined atmospheric transport distances of PAHs and also PCBs. In general, polycyclic aromatic hydrocarbons and PCB's tend to decline with distance from urban areas due to lower emission rates. However, persistent gas-phase PAHs, often alkylated PAHs, increase from urban to rural locations (Gingrich and Diamond 2001). These persistent gas phase compounds can travel as far as 50 km, whereas more reactive gas-phase and particle-phase compounds often travel shorter distances, <5 km (Gingrich and Diamond 2001). The impacts of atmospheric transport of PAHs can be seen at the urban fringe, in areas that are undergoing rapid urban spawl, where rapid increases in PAHs in lake sediments have been linked to increased automobile use for commuting (Van Metre et al. 2000). These findings suggest that urban sprawl may adversely impact water quality within a watershed due to large increases in traffic to and from urban centers.

Like PBCs and DDTs, polycyclic aromatic hydrocarbons have also been strongly correlated to the percent of commercial, industrial and transportation (CIT) land use in a watershed (Figure 9.3). Based on their land use impact model from the Northeast US, Chalmers et al. (2007) predict that PAHs will increase by a factor of 6 when a rural site becomes suburban and increase by a factor of 5 when the suburban sites becomes urban. Their model also predicts that the PEC for PAHs will be exceeded at 13% CIT. Research in other regions of the US is warranted but this study provides evidence for upward trends in PAHs with land use intensification. Based on these findings and the upward trend in PAHs in urban lake sediment cores relative to reference lakes (Van Metre and Mahler 2005), it is expected that PAHs will likely to surpass chlorinated hydrocarbons as a threat to human health and aquatic biota in streams and lakes in the coming decades.

Polycyclic aromatic hydrocarbons are emerging as threats to human and aquatic health, and the sources of PAHs include atmospheric deposition from incomplete combustion of fossil fuels as well as watershed sources such as asphalt sealants. Reducing atmospheric sources of PAHs such as vehicular traffic remains challenging and will require transportation alternatives and stricter zoning controls on urban sprawl. On the ground, planners, watershed managers, developers, and individual homeowners can apply best management practices to reduce watershed sources of PAHs by reducing or eliminating driveway sealants and/or finding alternative sealants. Structural practices described above can also reduce delivery of PAHs to streams.

Other organic compounds. Many organic compounds such as pharmaceutical, hormones, and other organic wastewater compounds are not currently regulated by the US EPA and cannot be readily removed by wastewater treatment or septic systems. A recent national survey of 139 US streams revealed that organic contaminants including pharmaceutical, hormones, and other organic wastewater compound (OWC) were detected in 80% of the rivers sampled (Kolpin et al. 2002). Most frequently detected compounds included coprostanol (fecal steroid), cholesterol (plant and animal steroid), N,N-diethyltoluamide (insect repellant), caffeine (stimulant), triclosan (antimicrobial disinfectant), tri(2-chloroethyl)phosphate (fire retardant), and 4-nonylphenol (nonionic detergent metabolite) (Kolpin et al. 2002). The impact of these individual compounds and the interaction of them on aquatic and human health remain unclear. Further research is warranted to understand the aquatic and human health risks of these compounds.

Nutrients

Nutrient concentrations have increased in rivers throughout the US and the world (Howarth et al. 1996; Mueller and Spahr 2006). Non-point sources of nitrogen (N) and phosphorus (P) dominate surface waters (Howarth et al. 1996; Carpenter et al. 1998; Caraco and Cole 1999) and are highly correlated with population density and net anthropogenic inputs to the watersheds. Dominant sources of nitrogen (N) to watersheds include fertilizers, atmospheric N deposition, food and animal feed imports, and biological N fixation in leguminous crops. Because P and sometimes N can limit productivity of surface waters, one of the main impacts of nutrients is through the process of eutrophication whereby lakes, reservoirs and sometimes rivers have excess algal or plant growth leading to degradation of water bodies. High levels of nitrate (>10 mg/L nitrate-N) in surface and ground waters can also have human health consequences, interfering with the ability of the blood to carry oxygen, particularly in infants.

A growing body of research indicates that nutrient concentrations and loads in rivers increase as development increases. In a national survey of streams and rivers, Mueller and Spahr (2006) showed concentrations of all nutrients (total nitrogen, total phosphorus, nitrate, orthophosphate) were significantly greater in partially developed watersheds than ing undeveloped watershed but significantly less than more developed agricultural, urban, and mixed land-use watersheds. Other studies have shown similar patterns at smaller spatial scales (Groffman et al. 2004; Lewis and Grimm 2007). The impact of exurban development on nutrient loading to rivers is most apparent in mountainous headwater catchments undergoing rapid development, such as those in Colorado. In these systems, exurban development has been linked to increases in dissolved inorganic nitrogen in streams during spring melt; 19-23% of this nitrogen is estimated to be coming from septic systems (Kaushal et al. 2006). Increased nitrate export from these headwater catchments indicates that the biotic capacity of these headwater catchments to take up this N is very limited, indicating these systems are more sensitive to development than other systems. Kaushal et al. (2006) suggest that because much of the world's population relies on water originating from mountain areas, even modest levels of nutrient enrichment can result in cascading effects on water supplies downstream.

Metals

Trace metals are metals found in very low concentrations in nature, and include arsenic (As), silver (Ag), cadmium (Cd), chromium (Cr), copper (Cu), mercury (Hg), iron (Fe), manganese (Mn), nickel (Ni), lead (Pb), zinc (Zn) and others. Several of these metals are essential for plant and animal life at low concentrations but become toxic at higher concentrations (Miller and Miller 2007). Trace metals can enter terrestrial and aquatic ecosystems by atmospheric deposition and point and non-point sources. Atmospheric deposition is an important source of trace metals, particularly for Hg and Pb. Indeed, some studies suggest that nearly all the Hg in water, sediment and fish can be explained by atmospheric deposition (Sorensen et al. 1990). Point and non-point sources of trace metals include mining byproducts, chemical waste, coal and industrial waste, metal plating, and plumbing.

Like organic compounds and nutrients, trace metals are highly correlated with population density and percent commercial, industrial, and transportation land use (CIT) (Figure 9.3). For example, the sum of trace elements (Cu, Pb, Hg, Zn) in streambed sediments is highly correlated with population density (Rice 1999), and anthropogenic Pb and Zn concentrations can be accurately predicted by population density (Callender and Rice 2000). However, spatial distribution of Pb and Zn depends on time of development; removal of leaded gasoline in the late 1970's has led to declines in lead and increased vehicular usage has kept Zn concentrations high in runoff and sediments. Consistent with these spatial observations, declines in lead have been observed in long-term lake sediment records, whereas high Zn concentrations have been observed in recent exurban developments because of high vehicular traffic (Mahler and Van Metre 2006). Work by Chalmers et al. (2007) suggests that metal concentrations will double when a rural site becomes suburban and again triple when the suburban site becomes urban (Table 9.1, Figure 9.3). Finally, concentrations of metals are predicted to exceed PEC standards between 3% CIT for Pb and 10-25% CIT for Zn, Hg, Cu, and Cd. Best management practices for metals include reducing source of metals by reducing vehicular use and traffic. Elimination or use of alternative chemicals for timber treatment and preservation that do not contain metals will also reduce metals in watersheds. In addition, structural practices to reduce delivery of metals to streams can be implemented including, but are not limited to, LID methods. Finally, more transformative planning options to reduce metals in watersheds include transferring development rights in undeveloped watersheds to more developed watersheds that are already likely to be already impacted by metals and other contaminants.

Sedimentation

Numerous studies have linked in-stream sediment conditions to upland landscape components and land-use change in watersheds (e.g. Richards et al. 1996; Wohl and Carline 1996; Sutherland et al. 2002; Opperman et al. 2005). Agricultural and urban land use activities are often considered the key drivers increasing fine sediment production and delivery to streams (Waters 1995; Pimentel and Kounang 1998). Compared to land with native vegetation, agriculture can result in significantly higher rates of sediment production, even on moderate slopes, due to the increased amount of bare soil exposed to the erosive power of raindrops and sheet wash (Dunne and Leopold 1978; Chang et al. 1982; Pimentel and Kounang 1998) and indirectly due to higher rates of runoff that lead to increased incision and bank erosion (Chang et al. 1982). Urban areas produce large amounts of fine sediment during periods of construction and can continue to produce sediment over longer time periods from bank erosion (Trimble 1997; Pizzuto et al. 2000). Less is known, however, about the impacts of exurban development and the relative impacts of different land uses on sediment production and delivery to streams; this issue is the focus of the watershed-scale case study described below.

A summary of a limited set of studies examining the impact of different urban land use types on water quality indicate that the process of urbanization dramatically increases the occurrence of organic pollutants as well as nutrients, metals, and sediments in streams (Table 9.1). In most cases, conversion of rural areas to suburban development is predicted to double the concentration of contaminants, with the exception of PAHs that is predicted to increase by 6 fold. Conversion of suburban to urban is predicted to result in a 3 to 5 fold increase in contaminants. Based on Chalmers et al. (2007) study, once the percentage of commercial, industrial and transportation land use exceeds 15%, contaminants in sediments will likely exceed the Probable Effect Concentration (PEC) and adversely impact aquatic and possibly human health. Further studies are warranted in other regions of the US to evaluate these relationships, and several studies are ongoing in the rapidly urbanizing desert Southwest in Tucson and also Phoenix. However, the study by Chalmers et al. (2007) suggests that the process of urbanization, the conversion of undeveloped land to suburban and ultimately to urban land uses, dramatically increases the occurrence of organic pollutants, nutrients, and metals in the environment and reduces water and sediment quality.

To reduce these contaminants, planners and watershed managers should implement best management practices. One the most effective means to reduce these contaminants in watersheds is to eliminate use or not introduce these contaminants in the first place. For example, developers and homeowners can potentially eliminate PAH-containing driveway sealants and advocate for PAH-free alternatives. Structural practices can help to reduce the delivery of contaminants to streams once they are introduced into the environment. Finally, there are more transformative planning options available to planners, conservationists, and watershed managers. One of these options, which we describe below in detail, involves the development of coupled land-use impact-land use change models and use of Transfer of Development Rights (TDR) to protect sensitive land by transferring the "rights to develop" from one area and giving them to another.

ASSESSMENT OF RELATIVE LAND-USE IMPACTS AND APPLICATION OF COUPLED ENVIRONMENTAL -ECONOMIC LAND-USE CHANGE MODEL TO WATERSHED PLANNING: A CASE STUDY FROM THE RUSSIAN RIVER BASIN, CALIFORNIA

Overview of Coupled Environmental-Economic Land Use Change Model

We draw on our research from the Russian River Basin in Sonoma County, California to demonstrate how the steps outlined in Table 9.2 are used to develop the necessary models needed to project changes in water quality with future land use change (Figure 9.1). Specifically, we analyzed the relative impacts of three different land uses (urban, exurban, and vineyard development) on levels of fine sediment in streams (Lohse et al. 2008). Fine sediment is one measure of water quality that reduces habitat suitability for spawning salmonids, and data on levels of fine sediment or embeddedness were available throughout much of the Russian River basin from the California Department of Fish and Game. We used these data to quantify the impacts from urban and agriculture on elevated levels of fine sediment in streams, as well as determine the extent to which exurban development impacts water quality (Step 1). We then developed an economic land use change model (Step 2) and land price valuation model (Step 3). Because all of our models utilized parcel-level data, we could distinguish among low, medium, and high-density housing, detect relative effects of these different types of housing as well as other land uses on watershed processes, resolve geographic resolution issues between models, and couple these models to forecast expected loss of water quality and assess the risk of land-use conversion at a watershed scale (Step 4). Finally, we could compare these expected losses in water quality to the cost of preventing these impacts through upland conservation. This case study provides an important demonstration of how coupling environmental, land-use, and economic modeling can provide a useful tool to forecast scenarios that examine environmental and economic trade-offs and can help guide land-use planning for conserving and restoring water resources.

Step 1: Environmental Data for Land-Use Impact Model

The first step of our integrated modeling approach was to develop an empirical land-use impact model using land-use classification data at the parcel-level scale and water quality on levels of fine sediment from low (1) to high (4) for each sampled stream reach. We classified land use using parcel data from the county assessor's office to provide more accurate residential classification than LANDSAT TM imagery. A comparison of land-use classification by parcel-level data and LANDSAT TM imagery in Figure 9.4 shows how much better the former source data is for mapping exurban development. In our case, the 1997 Sonoma County tax-assessment data contained the number of residential structures, year built, lot size, and other land-use characteristics for each parcel. We distinguished between urban (<1 acre per structure) and rural-residential (1-40 acres per structure) because a housing density of 1 acre per structure is the typical limit on residential development serviced by septic systems (Newburn and Berck 2006). Intensive agriculture in these watersheds was almost exclusively wine grape growing. Parcels were classified as vineyard if the parcel had at least 10% vineyard or at least 5 hectares of vineyard based on digital maps derived from aerial photographs.

We then used ordinal logistic statistical modeling techniques to develop response models relating the rank level of fine sediment at each salmonoid spawning site with land-use and other independent variables (see Hosmer and Lemeshow 2000 for details on model development). The field data on fine sediment levels was included from 93 watershed reaches with an average of 54 spawning sites per reach. To examine the relationship between land use and level of fine sediments surrounding spawning gravels, as a measure of spawning-habitat quality, we used a 10-m Digital Elevation Model in a Geographic Information System (GIS) to delineate watersheds above the downstream end of all the surveyed reaches that met several sampling criteria. Explanatory variables included existing aerial percentages of vineyard, urban, and rural-residential land use in 1997 within each upstream watershed. Biophysical watershed variables such as road density, a stream power index, a hillslope stability index as well as categorical variables including channel type, dominant geology, and river bank-substrate material (bedrock, boulder, silt/clay, cobble) were also included.

The results of the ordinal logistic model showed fine sediment levels decreased with increasing percentage of three land-use types (urban, exurban and vineyard) in watersheds, with urban development having a larger marginal impact than either rural-residential or vineyard use (Lohse et al. 2008). To explore the relative importance of including exurban development in our model, rather than just urban development as is done in most models, we compared statistical models with and without exurban land-use. We found our predictions of fine-sediment were better when the amount of exurban development was included. To examine the robustness of the model, we performed a goodness of fit test in which we included an additional data set of watersheds in the statistical model that were not used for model building. The coefficients in the model were not significantly altered by this additional data set suggesting a robust model fit.

Step 2: Land use change modeling

The next step of the process was the construction of a land-use change model using parcel data for the period 1994-2002 using the parcel-level land-use data described above. Land-use conversion was defined as any transition from developable parcel into vineyard, urban or exurban development during the period 1994–2002. A multinomial logit model, a statistical model that predicts the probability of an event occurrence by fitting categorical and/or numerical data to a logistic curve, was developed to explain land-use transitions as a function of parcel-site characteristics, including average slope, growing degree days (microclimate), 100–year floodplain, proximity to major employment centers, designated sewer and water services (SWSA), and minimum lot-size zoning.

The estimated coefficients from the multinomial logistic regression were used to predict the site-specific conversion probabilities for each developable parcel based on mapped site characteristics. Then we used the land-use change model to simulate the estimated future amount and location of development for each land-use type within each watershed. During this build-out step, each parcel could remain developable or become converted to one of the five developed land-use types in a given simulation according to the site-specific conversion probabilities. To map the forecasted future pattern of land use, we added the amount of estimated land-use change to the actual extent of land-use type already mapped. Our future development scenarios spanned 2002–2010 because the land-use change model was based on development over an eight year period (1994–2002). In our case, we forecasted the amount of land-use change under a "business-as-usual" scenario; however, other scenarios including changes in land-use policies could be run as well.

Step 3: Economic Valuation Model

To examine the economic tradeoffs along with expected changes to water quality, we employed an existing parcel-level land valuation model. The estimated land values were derived from a hedonic price model that uses recent property transactions to estimate the market value of developable land as a function of site-specific characteristics (Newburn et al. 2006). Tax Assessor data provided the necessary information on individual parcels for the land value, current land use, and other property characteristics. We then used a similar set of explanatory variables for each parcel, including characteristics for land quality (slope, elevation, microclimate, 100-year floodplain), accessibility (travel times to urban centers, sewer and water service), neighboring land-use externalities (percent protected open space and urban), and zoning (land use designations, minimum lot size) to build a spatially explicit economic land valuation model. For each developable parcel, we were then able to estimate the value of developable land because key site characteristics were known within the GIS. As shown in Figure 9.5, the predicted value of developable land was observed to range over several orders of magnitude. The large degree of variation in land prices highlights why priority setting for land conservation should include the spatial heterogeneity in land values (Figure 9.5).

Step 4: Environmental- economic decision support tool

The final step of coupling the land-use impact model and economic land use change model involved forecasting the probability distribution of fine sediment levels based on the expected percentages of each land-use type in 2010 and the estimated parameters in the ordinal logistic model. In our case study, we found that forecasted rural-residential and vineyard development had much larger influences on decreasing water quality than urban development. This is because the land-use change model estimated 10 times greater land-use conversion to both rural-residential and vineyard compared to urban. Also, forecasted urban development was concentrated in the most developed watersheds, which already had poor spawning substrate quality, such that the marginal response to future urban development was less significant.

A land conservation targeting rule was used to identify priority areas for protection based on the expected loss of water quality due to future land-use conversion and the average land costs in each watershed. In our case, we maximized conservation goals based on the objective of minimizing the expected loss in environmental benefit per unit cost (Newburn et al. 2006). Applying this targeting rule to our results, we identified priority areas by summing the relative probabilities of loss of high quality sites (levels 1 and 2) and dividing by the average cost per acre for that watershed (Figure 9.6). Figure 9.6 shows the initial level of development in the watersheds and the change in water quality with future land use change. Findings from our study suggested that conservation efforts should target moderately and less-developed watersheds to meet the goal of protecting water quality for salmonid-spawning habitat, where future rural-residential development and vineyards threaten high-quality fish habitat, rather than the most-developed watersheds, where land prices typically are much higher and land-use development, particularly urban development, has already resulted in significant habitat degradation (Figure 9.5). With this decision support tool and spatial maps of environmentally sensitive watersheds at risk to future development, decision-makers can work to influence the density and location of future residential development through local zoning and other land-use policies. In particular, we recommended that a Transfer of Development Rights (TDR) program be implemented to curtail lower-density rural-residential development within moderate- and less-developed watersheds (sending areas in light grey, Figure 9.6), while encouraging higher-density infill urban development to take place in areas already highly disturbed (receiving areas in dark grey watersheds, Figure 9.6). Finally, in concert with these more transformative planning tools, effective runoff and construction control techniques, best management practices for road construction and maintenance (*public communications*, Roads, Highways, Bridges-NPS categories, http://www.epa.gov/owow/nps/roadshwys.html), and other low impact development (LID) strategies (*public communications*, Low-Impact Development, www.EPA.gov/owow/nps/lid/) can be used at a local scale when development does occur in sensitive watersheds.

CONCLUSIONS

In this chapter, we examined the impacts of urbanization on water quality, with particular emphasis on the expansion of housing development beyond the urban fringe. A review of the literature indicates that conversion from rural to suburban increases the occurrence of organic pollutants, as well as nutrients, metals, and sediments in streams, with potentially adverse effects for stream biota and humans. The main drivers behind these patterns appear to be increased population density, increased roads and hence increased vehicular traffic that results in deposition and fluvial transport of metals and organic pollutants to streams and sediments. Exurban development and associated roads also extend possible land-use impacts by increasing nutrient and fecal bacteria inputs to streams through leaking septic systems and increased vehicular traffic, both of which result in increased nitrogen inputs to watersheds.

We used a case study from the Russian River Basin in California to illustrate how to couple water-quality modeling and land-use change models to guide conservation planning and development. In this case study, we developed a land-use impact model to predict the relative impacts of urban and exurban development on sedimentation in streams that support rare salmonids. We then coupled this model with a land-use change model to predict where conversion to exurban and urban development would take place (Figure 9.6). With both models, we differentiated between exurban and urban because they represent different types of growth, and we showed that they have differential impacts on substrate quality in streams. We showed that exurban development has a larger potentially impact on water quality because of its ability to "leapfrog" into previously undeveloped areas, a finding that is likely to transfer to different environmental settings.

The case study presented here demonstrates that land-use impact modeling can be coupled with land-use change and economic modeling to forecast scenarios of future ecological conditions and therefore guide land-use and restoration planning. Although we are unable to generate highly accurate predictions, we need to examine the inherent trade-offs associated with various policy options to (1) better inform local decision-makers about trade-offs and (2) identify options that result in water and land conservation. The importance of scenario planning and engaging communities in this process is widely recognized (Hopkins and Zapata 2007), and we argue that our ability to integrate conservation planning with land-use planning hinges on our ability to develop useful future scenarios. Given that there are always multiple ecosystem services traded-off for any proposed ecosystem alteration, it is critical that scenarios be developed that take uncertainty into account and reduce the changes for unintended consequences or perverse results. We conclude that coupled environmental impact-land use change models provide an adaptive-management tool to manage existing threats to surface and ground water quantity and quality as well as plan for future impacts owing to changes in land use.

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Kathleen Lohse

Kathleen Lohse is an ecosystem scientist who joined the faculty of the School of Natural Resources at the University of Arizona in January 2007. She received her Ph.D. from the University of California Berkeley in 2002 in Soil Science with an emphasis on Ecosystem Science and B.S. and B.A. in 1993 in Urban and Regional Studies and Biology from Cornell University. She works at the interface of ecology, earth system science and land-use planning, studying the processes shaping watersheds and their responses to anthropogenic changes. Her primary research interests include examining the effects of human activities such as nitrogen (N) deposition and land use changes on soil properties and hydrologic transfer of nutrients and sediments to downstream ecosystems, determining the environmental consequences of these alterations for river- riparian ecosystems, exploring the interactive controls of vegetation change, management practices, and fire on ecosystem and geomorphological processes, and coupling spatially explicit biophysical models with land use change models to predict cumulative watershed effects. Her current research examines the possible tradeoffs between enhanced urban runoff-recharge and water quality in the Tucson Basin, Arizona.

Adina Maya Merenlender

Adina Merenlender is on the faculty at University of California, Berkeley (Environmental Science, Policy and Management Department) and is an internationally recognized conservation biologist working on environmental problem solving at the landscape-scale. She has published more than 65 scientific research articles focused on underlying relationships between land use and biodiversity and recently co-authored the only comprehensive book on wildlife corridor planning, "Corridor Ecology: The science and practice of linking landscapes for biodiversity conservation." Over the past fifteen years she has trained graduate students as well as worked with decision-makers to address the forces that influence loss of biodiversity including the use of spatially-explicit decision-support systems. She is a member of the North American Board for the Society of Conservation Biology and on the editorial board of Journal for Conservation Planning.

Figure Legends

Figure 9.1: Development of coupled biophysical-land use change model. Steps include (1) quantifying relationships between land use and water quality/quantity, (2) land-use change scenarios that forecast likely future land-use change scenarios, 3) economic valuation models, and 4) integration of these models to evaluate environmental and economic trade-offs.

Figure 9.2: A) Percent of streams impaired by different sources of contaminants or alternations including pathogens, siltation, habitat alteration, oxygen depleting substances, nutrients, metals, and flow alteration (adapted from the USEPA 2000); B) Percent of sediments contaminated by organics, PCBs, pesticides, PAHs, metals, and mercury (adapted from (US EPA 1997)).

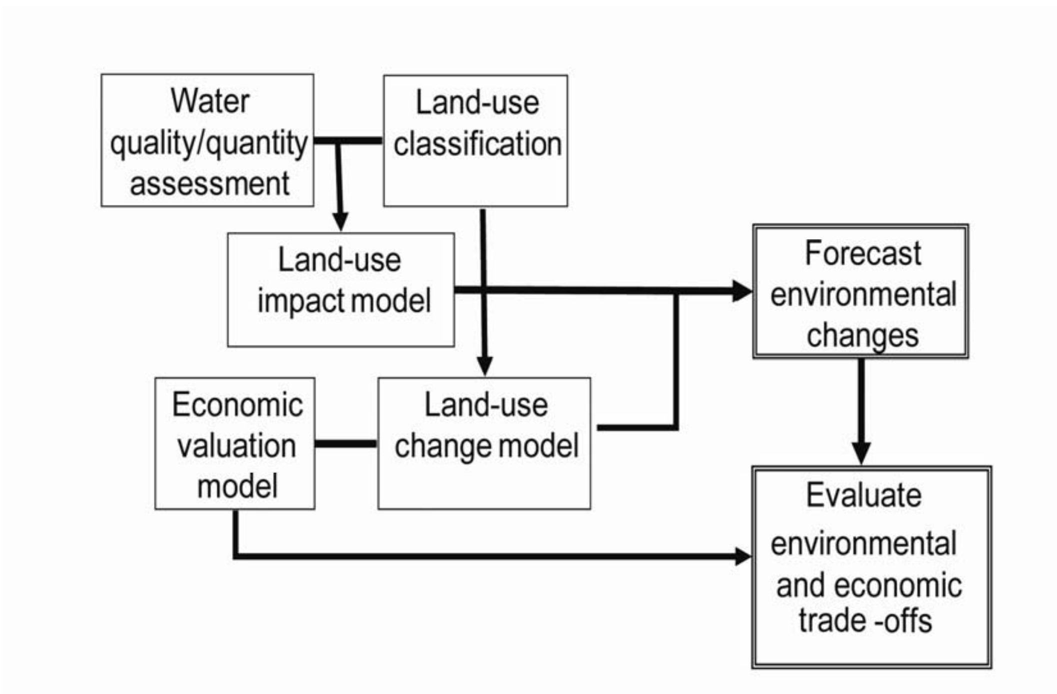
Figure 9.3: Impact of % of watershed in commercial, industrial and transportation (CIT) on metals and organic compounds in sediments in the Northeast US. Red line indicates the Probable Effect Concentration (PEC) and the dashed line indicates the reference concentrations. Regression lines are shown as black solid lines and CIT threshold when CIT exceeds PEC is indicated by the intersection of the red and black line (adapted from Chalmers et al. 2007).

Figure 9.4: Comparison of parcel-level and LANDSAT TM imagery land use (based on CalVeg classification) in the study region in Sonoma County, California. Parcel-based, land-use classification for 1997 with rural-residential development (1 to 40 acres/structure) and urban development (<1 acre/structure). Residential densities were based on parcel records obtained from the Sonoma County tax assessment office. Vineyard land use was digitized from aerial photos in 1997.

Figure 9.5: Initial conditions of land-use development in the study watersheds and cost per parcel area.

Figure 9.6: Change in probability distribution of spawning substrate quality with forecasted land-use change and probability of loss of high-quality substrate/ land cost in least to most developed watersheds in the Russian River Basin. The watersheds colored light grey represent the lowest cost option for conserving high-quality spawning habitat.

Figure 9.1



Water quality/quantity assessment
Land-use classification
Land-use impact model
Forecast environmental changes
Economic valuation model
Land-use change model
Evaluate environmental and economic trade -offs

Figure 9.2

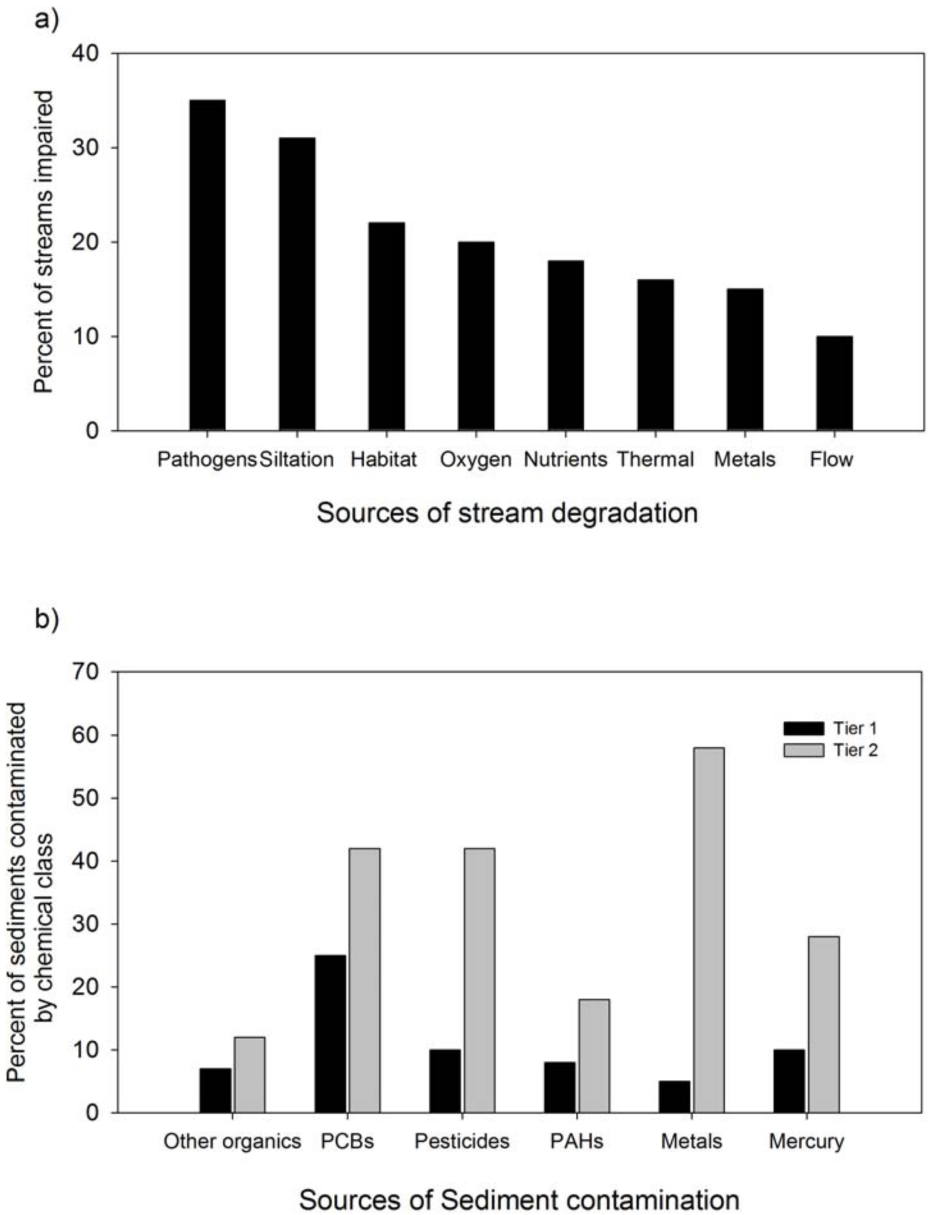
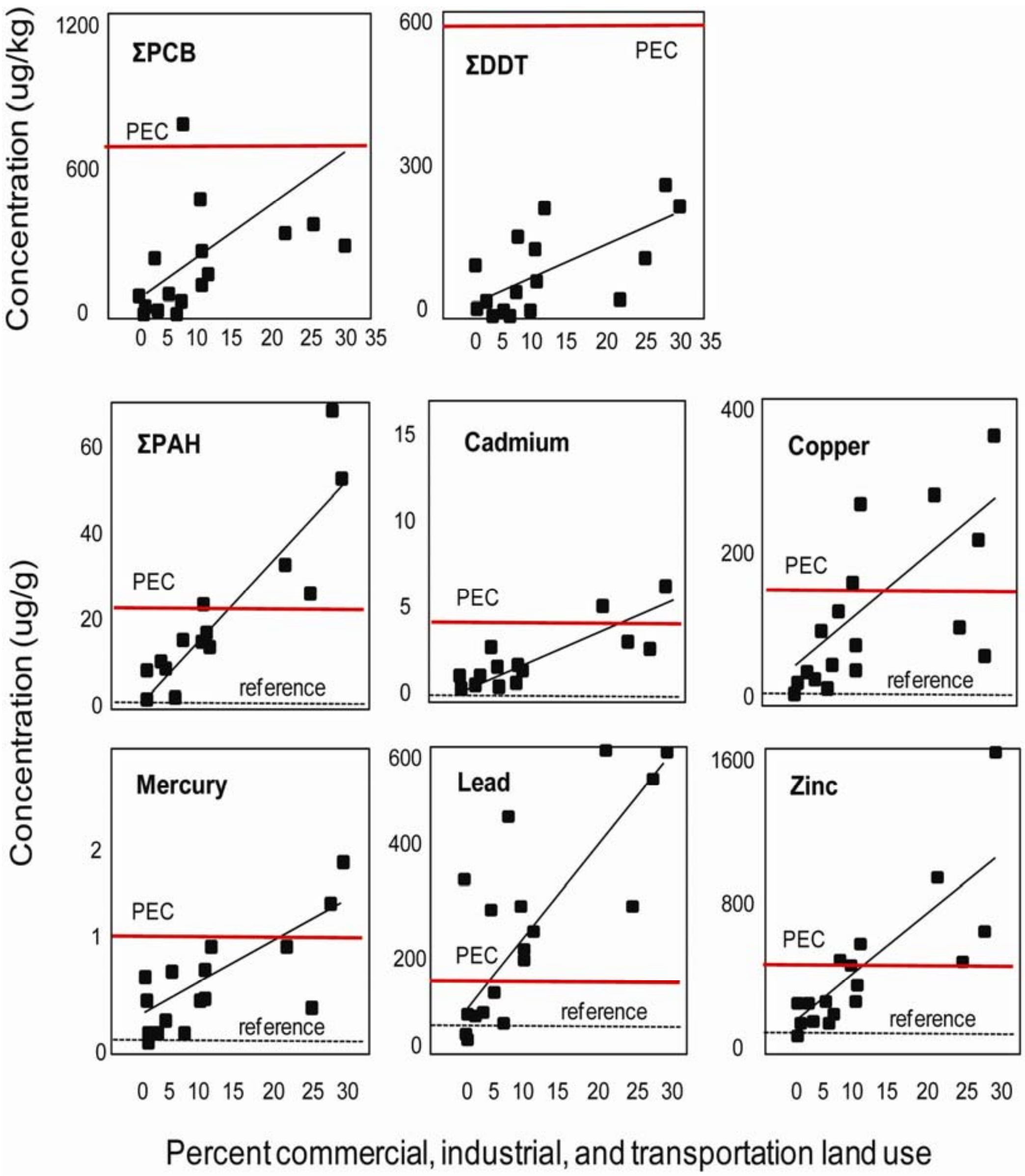
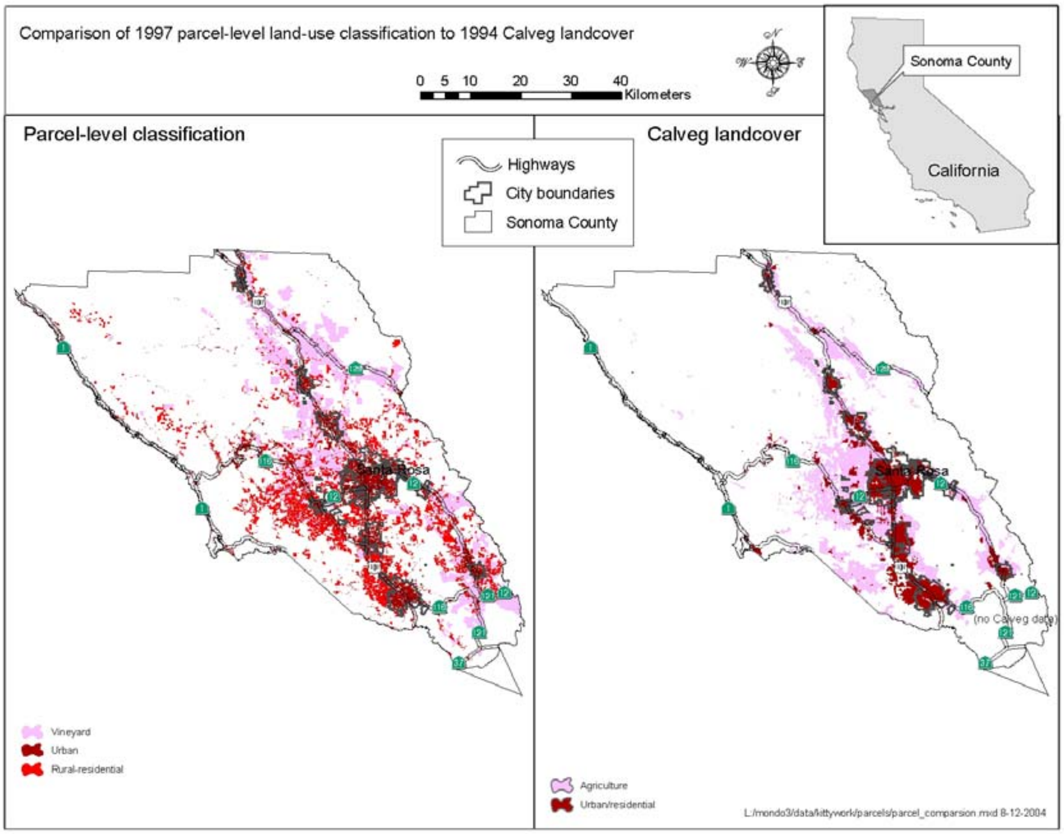

a)
40
30
20
10
0
Percent of streams impaired
Pathogens
Siltation
Habitat
Oxygen
Nutrients
Thermal
Metals
Flow
Sources of stream degradation
b)
70
60
50
40
30
20
10
0
Percent of sediments contaminated by chemical class
Tier 1
Tier 2
Other organics
PCBs
Pesticides
PAHs
Metals
Mercury
Sources of Sediment contamination

Figure 9.3



Concentration (ug/kg)
ΣPCB
PEC
1200
600
0
0 5 10 15 20 25 30 35
ΣDDT
PEC
600
300
0
0 5 10 15 20 25 30 35
Concentration (ug/g)
ΣPAH
PEC
reference
60
40
20
0
Cadmium
PEC
15
10
5
0
Copper
PEC
reference
400
200
0
Mercury
PEC
reference
2
1
0
0 5 10 15 20 25 30
Lead
PEC
reference
600
400
200
0
0 5 10 15 20 25 30
Zinc
PEC
reference
1600
800
0
0 5 10 15 20 25 30
Percent commercial, industrial, and transportation land use

Figure 9.4



Comparison of 1997 parcel-level land-use classification to 1994 Calveg landcover
0 5 10 20 30 40
Kilometers
Sonoma County
California
Parcel-level classification
Calveg landcover
Highways
City boundaries
Sonoma County
Santa Rosa
(no Calveg data)
Vineyard
Urban
Rural-residential
Agriculture
Urban/residential
L:/mondo3/data/kittywork/parcels/parcel_comparsion.mxd 8-12-2004

Figure 9.5

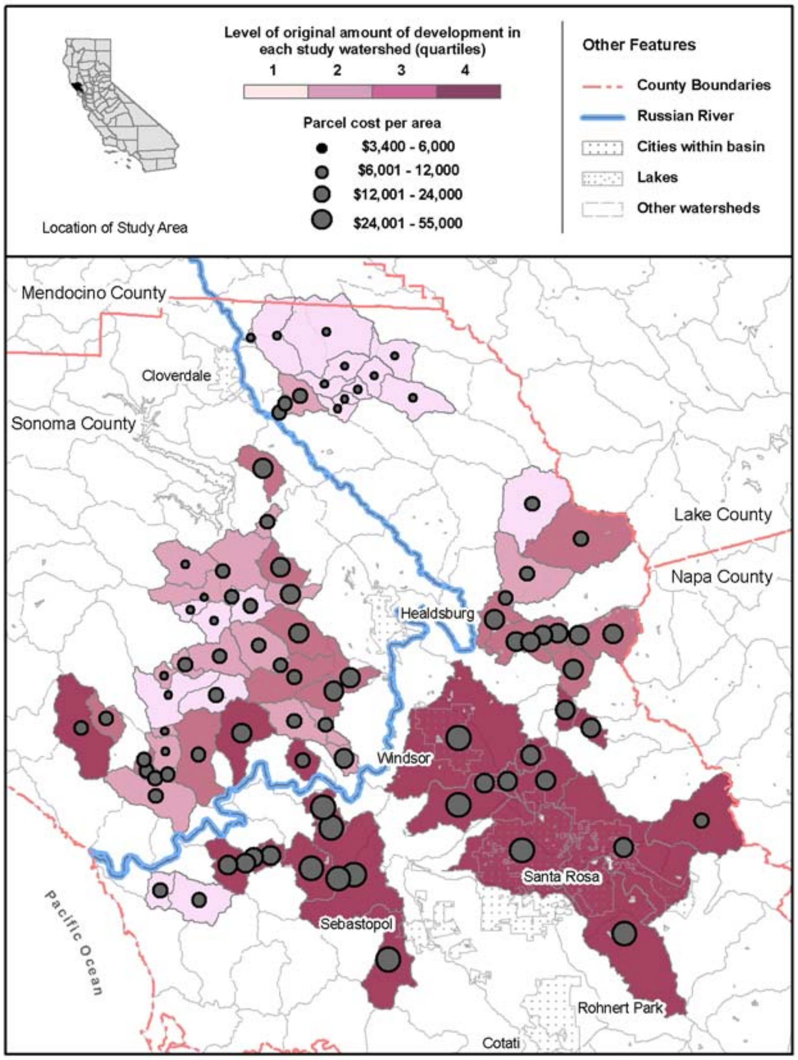
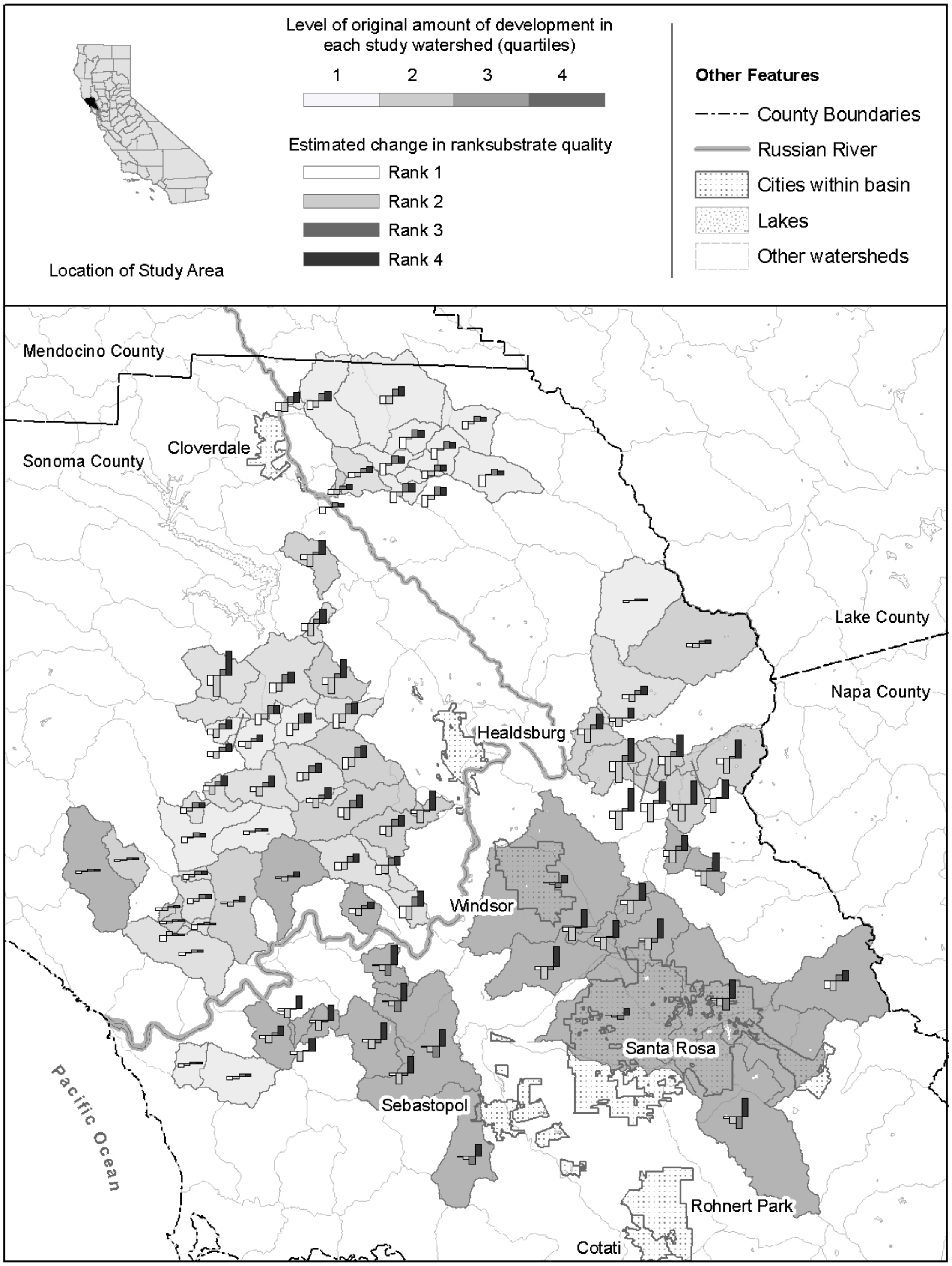

Level of original amount of development in each study watershed (quartiles)
1
2
3
4
Parcel cost per area
$3,400 - 6,000
$6,001 - 12,000
$12,001 - 24,000
$24,001 - 55,000
Location of Study Area
Other Features
County Boundaries
Russian River
Cities within basin
Lakes
Other watersheds
Mendocino County
Cloverdale
Sonoma County
Lake County
Napa County
Healdsburg
Windsor
Santa Rosa
Sebastopol
Pacific Ocean
Rohnert Park
Cotati

Figure 9.6



Level of original amount of development in each study watershed (quartiles)
1
2
3
4
Estimated change in ranksubstrate quality
Rank 1
Rank 2
Rank 3
Rank 4
Location of Study Area
Other Features
County Boundaries
Russian River
Cities within basin
Lakes
Other watersheds
Mendocino County
Sonoma County
Cloverdale
Lake County
Napa County
Healdsburg
Windsor
Santa Rosa
Sebastopol
Pacific Ocean
Rohnert Park
Cotati

Table 1: Contaminants, sources of organic pollutants, nutrients, and metals, Probable Effect Concentration (PEC) for sediments (in ug/g or ug/kg sediment) or Maximum contaminant level (MCL) for drinking water (in mg/L), and predicted enrichment factor (EF) from conversion from rural (R) to suburban (S) to urban land use (U). Percent commercial, industrial, and transportation (CIT) or impervious surface area (ISC) is used as an index of urban intensity. Drinking water standards are specified because regional water quality standards vary for rivers and streams (from Chalmers et al. 2007).

Contaminants		**Sources**	**PEC or MCL**	**EF (R to S)**	**EF (S to U)**	**Threshold CIT or ISC > PEC or EPA**
Organic Pollutants						
Pathogenic	*Escherichia coli (E. coli)* Enteric bacteria Viruses	Septic, domestic animals, wildlife, or agriculture	0 MCL 400 MPN/ 100ml*			10-20% ISC*
Toxic organic	Polyaromatic hydrocarbons (PAH)	By products of fossil fuel combustion in automobiles, power plants, and heating facilities. Creosote and roofing tar Coal-tar and asphalt sealant	22,800 ug/kg	6.1	5.2	13
	Halogenated					

	hydrocarbons					
	PCB	Solvents, cleansers, and degreasers but also pesticides, electrical equipment, and commercial products	676 ug/kg	2.2	3.7	31
	DDT	Pesticides Insectides	572 ug/kg	1.8	3.2	Not exceeded
Inorganic contaminants						
Nutrients						
	Nitrate-N	Fertilizers, atmospheric Deposition, food import , waste	10 mg/L (MCL)			
	Nitrite-N		1 mg/L (MCL)			
	Total phosphorus	Fertilizers	-			
Metals						
	Arsenic (As)	New/old timber treatment, ag. chemicals and natural sources; mining by product	33,000 ug/kg	-	-	-
	Cadmium (Cd)	Tires, tire wear and galvanized metals	5 ug/g	1.9	3.3	25
	Chromium (Cr)	Metal plating	111 ug/g	-	-	-
	Copper (Cu)	Break pads,	149	1.7	3.1	14

		residential algaecides, wood preservatives, landscaping materials	ug/g			
	Mercury (Hg)	Industrial waste, mining, fuels	1.06 ug/g	1.7	3.1	23
	Lead (Pb)	Old housing (leaded paint and fuel)	128 ug/g	1.9	3.4	3
	Zinc (Zn)	New urban surfaces, rooftops, wood preservative	459 ug/g	2	3.5	10
Sediments		New construction, roads, etc				

*for water quality criterion in Georgia (Schoonover and Lockaby 2006).

Table 2: Steps to develop coupled environmental impact model and land use change model for guiding decision making, planning, ecosystem management, and conservation

Modeling components	Steps
1. Environmental data for environmental response modeling	a. Collect spatial data on water quality/quantity/habitat b. Delineate scale of analysis in-line with scale of land use data c. Select statistical model to examine potential causal relationships d. Perform variable selection e. Select best models f. Test resulting models on an alternate data set
2. Land use change modeling	a. Acquire land-use data at a parcel scale b. Consolidate data into relevant land-use categories c. Derive land use variables d. Calculate other relevant variables
3. Economic valuation modeling	a. Develop empirically based valuation model

	using high resolution data b. Hedonic price modeling can be used to estimate market values c. Ensure the range of variability in relative value is represented
4. Ecological and economic decision support tool	a. Examine areas of forecasted change in water quality b. Divide probability of environmental degradation by cost of protection c. Select solution sets that maximize returns to natural and human systems