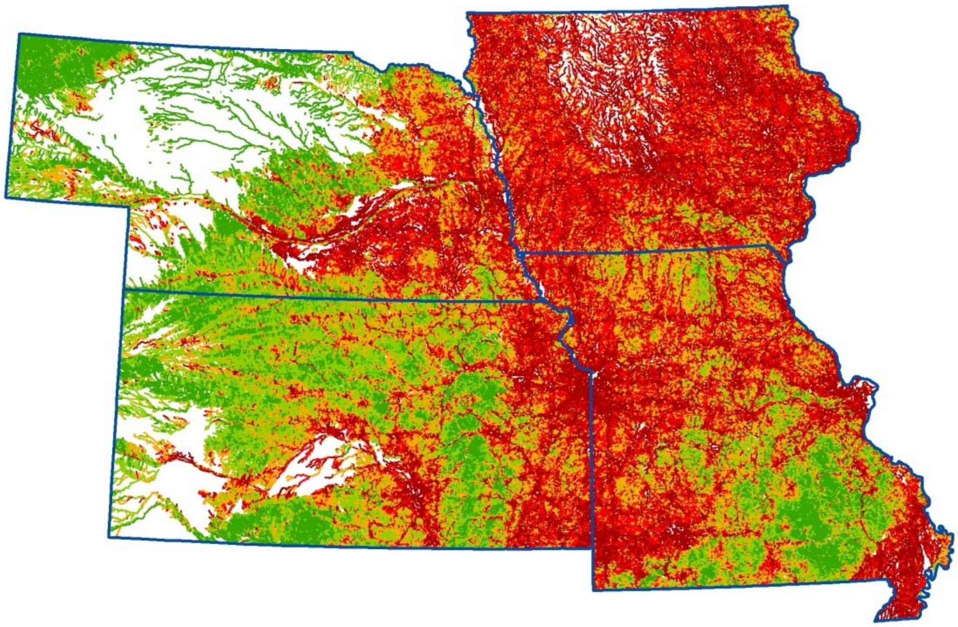
Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

Final Report

Revision May 2010



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Cover Photos: *Aaron Garringer*

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Executive Summary

There has been a tremendous amount of geospatial data developed over the last decade, much of which represents anthropogenic features on the landscape that have the potential to threaten the ecological integrity of streams. What has been lacking is a means of assembling, organizing, and quantifying this data in a fashion that will provide information relevant to each stream segment.

To help address this information gap the primary purpose of this project was to develop synoptic human threat indices for assessing the ecological integrity of freshwater ecosystems in EPA Region 7. To accomplish this, a large portion of this project consisted of data development in which we quantified threats in the drainage area above each stream segment in EPA Region 7 using a modified version of the 1:100,000 National Hydrography Dataset (NHD).

Approximately 40 geospatial data layers were identified through the help of a regional oversight committee that was brought together to provide guidance for this project and another related project for the state of Missouri. This committee consisted of individuals from each of the states comprising EPA Region 7 (Iowa, Kansas, Missouri, and Nebraska). A large thrust of this project was to collect and assemble data layers that were seamless across state boundaries. These seamless data layers allow information for watersheds that straddle state boundaries to have information quantified accurately for the drainage area above each stream segment. Once assembled and quantified these data were used to construct an index representing potential cumulative threat from anthropogenic activities to aquatic riverine ecosystem integrity. This index is referred to as the human threat index (HTI). The index was constructed by first creating a local human threat index (LHTI) and a watershed human threat index (WHTI). These two indices were subsequently combined into the overall HTI.

Threat quantification for the Missouri portion of EPA Region 7 was funded by a separate 319 grant and was designed to mesh seamlessly with the data quantified for this project. The Missouri project, that did not include the development of a "threat index", was completed in May of 2009 (MoRAP 2009). The final report for this Missouri work titled *Stream Reach Specific Watershed Data: Threats to Aquatic Ecosystem Integrity* was submitted to the Missouri Department of Natural Resources. Most of our methodology for gathering and quantifying threat datasets was developed jointly for both the Missouri project and this one. Because this information was described in detail in the Missouri report much of it will be repeated here.

Project Funding
The primary funding for this project which consisted of quantifying threats for the states of Iowa, Kansas, and Nebraska along with the development of a human threat index for every stream segment in EPA Region 7 (Iowa, Kansas, Missouri, and Nebraska) was provided by a U.S. EPA Wetlands Program Development Grant for EPA Region 7.

The U.S. Environmental Protection Agency Region 7, through the Missouri Department of Natural Resources, has provided partial funding Under Section 319 of the Clean Water Act for the quantification of threats in Missouri. The results of this project are presented in the report *Stream Reach Specific Watershed Data: Threats to Aquatic Ecosystem Integrity* was submitted to the Missouri Department of Natural Resources (MoRAP 2009).

Acknowledgements

We would like to thank all of the individuals who participated in one or more of our regional oversight committee meetings. These individuals provided invaluable insights, suggestions, and data. Without their help this project would not have been possible. Although too numerous to mention here their names and affiliations are listed in Appendix A. We would also like to thank Jeff Fore for providing insights into the methods used to create our final human threat index and Dyan Pursell for assistance with the literature reviews. Mike Morey was instrumental in developing processes and programs that enabled us to quantify distance to threats.

Finally, we would like to thank Tom Wilton (Iowa Department of Natural Resource), Ken Bazata (Nebraska Department of Environmental Quality), and Debbie Baker (Central Plains Center for Bioassessment) for providing fish collection data that was used to create seamless fish indices of biological integrity (IBI) over all of EPA Region 7. These data were use to help validate our human threat index.

Chapter 1

Introduction

The first step to effective resource management is having an accurate inventory of the, a) resources you intend to manage and b) factors that influence those resources (Fajen 1981). The use of geographic information systems (GIS) has certainly enhanced our ability to generate basic inventory statistics about natural resources and factors that might negatively influence these resources. Numerous geospatial datasets have been developed (e.g., geology, soils, land cover, streams, dams, mines) and these datasets have given resource managers the ability to develop inventories and conduct assessments that help guide the allocation of limited human and financial resources to those locations most in need of restoration or conservation.

Despite these advances in GIS and the increased availability of geospatial data there still exists a tremendous data gap with regard to freshwater resources. This data gap pertains to the fact that the physicochemical and biological character of a lake or particular stream reach is largely influenced by natural and land-use conditions within the watershed. While the United States Geological Survey (USGS)/Natural Resource Conservation Service (NRCS) Hydrologic Unit (HU) layers have somewhat helped with this problem, they do not provide a comprehensive picture of watershed conditions within the state since inventory data compiled for these HUs, only accurately characterize watershed conditions of the stream reach at the outlet of each HU. Even with the most detailed 12-digit HU coverage we can only accurately characterize the watersheds of a limited number of stream segments in EPA Region 7. This would represent a small percentage of the more than 400,000 individual stream reaches contained in the 1:100,000 National Hydrography Dataset (NHD) within Region 7.

As part of the Missouri Aquatic GAP Project, MoRAP developed a GIS methodology that can be used to quantify watershed and upstream riparian conditions for every single stream reach (between consecutive tributaries) within the 1:100,000 NHD. These data provide a powerful tool for developing comprehensive inventories and conducting detailed assessments for the freshwater resources within EPA Region 7. The reach-specific precision of these data allow inventories and assessments to move from a fixed unit state (i.e., HUs) to a continuum of data that provides the necessary flexibility to meet a wide range of research and management applications. For instance, we can now generate maps and linear statistics to display and quantify the number or percent of stream miles in EPA Region 7 that have greater than 10% (or any desired percentage) of their watershed draining urban lands, row crop agriculture, forest land, and other land uses.

When we consider natural resources management and think about what resource managers really do, we find that they don’t necessarily manage the resource itself, but often manage human activities that impact resource quality. Some common questions of resource managers include: What factors threaten the ecological integrity of a stream of interest? What threat is most pervasive? Where are these threats within the network or watershed? Answering these questions and others like them can help resource managers target specific threats at specific locations. Finally for a decision to be objective, it must be driven by data/information. More specifically, nearly all natural resource management decisions must be driven by spatially explicit (i.e., map-based) data/information.

In order to make effective decisions aquatic resource managers must have an understanding of the threats to aquatic ecosystem integrity. These threats may be local, residing at the stream reach of interest, or may be some distance upstream. The data developed as part of this project will help identify and quantify many of these threats in a high resolution, spatially explicit manner and provide an "index" that will shed insight into the degree of threat any given stream segment is experiencing in relation to all others. This index will help highlight areas that likely have diminished aquatic ecological integrity throughout EPA Region 7.

The goals and objectives of this project were:

Goal: Develop GIS-based Synoptic Human Stressor Indices that can be used as an initial screening tool for assessing the extent and specific causes of diminished ecological integrity of freshwater resources throughout EPA Region 7 at a fine resolution (e.g., stream reach).

Objectives:

1. Increase coordination among state, federal and local stakeholders in EPA Region 7 with regards to broad-scale assessments of the ecological integrity of freshwater resources.

2. Increase knowledge about the availability and limitations of existing geospatial data pertaining to human stressors and GIS-based methods of environmental assessment.

3. Develop a centralized repository of the most accurate and recent geospatial data pertaining to human stressors within EPA Region 7 in order to increase efficiency and standardization of efforts to document, prevent, reduce and eliminate water pollution.

4. Quantify the number, density, areal extent, and percentages for a variety of human land uses for every stream reach contained in the 1:100,000 NHD within Iowa, Kansas, and Nebraska. *Note: These data were developed for Missouri with monies from a separate 319 grant*

5. Develop scientifically rigorous, standardized, and regionally accepted synoptic indices that can be practically applied within a GIS and account for;
   a. local, watershed, and upstream riparian disturbances that negatively affect ecological integrity of surface waters, and
   b. differences in the magnitude and type of disturbance among all possible human stressors that are identified.

6. Calculate the resulting Synoptic Human Threat Indices for every stream reach contained in the 1:100,000 NHD within EPA Region 7.

7. Validate the accuracy of the resulting Synoptic Human Threat Indices.

We wanted to quantify potential human threats for the drainage area above each 1:100,000 NHD stream segment in EPA Region 7 and include human land-use factors for both nonpoint and point source pollution using existing data sets. In addressing these objectives we sought to create an aquatic 'threat assessment tool' that would be useful for on-the-ground planning and management. We wanted to utilize as many threat datasets as possible, consider the drainage area above each stream segment, incorporate riparian landcover, account for distance to upstream threats, and be seamless across state borders. Finally, we wanted to incorporate these cumulative threats into an overall human threat index (HTI).

To help us get started we utilized the expertise of members of a project oversight committee that was brought together to provide insights and guidance throughout the project. This committee consisted of approximately twelve active participants with representatives from state or federal agencies across the four states comprising EPA Region 7 (See Appendix A). The project's Regional Oversight Committee met five times.

A total of 35 individual threat attributes were quantified and are seamless across state borders. We also quantified distance to threat information for 15 of these data sets.

The resulting datasets developed for this project <u>will</u> help answer questions like: What threats are upstream? How much or how many threats are upstream? Where or how far are these threats upstream? The HTI will help shed light on ecological integrity and show the degree of impact any given stream reach is experiencing relative to all other stream reaches in EPA Region 7. The resulting datasets developed for this project <u>will not</u> answer questions like: What is the impact of a given threat upstream? Of the many threats upstream, which is worse? How exactly do these threats alter the physical/chemical character of the stream? Is there a threshold for possible impacts? What can be done to mitigate these potential problems? To help answer these questions resource managers will need to rely on addition information and their own areas of expertise.

Study Area

The study area for this project consists of EPA Region 7 which is comprised of the entire states of Iowa, Kansas, Missouri, and Nebraska (Figure 1). These four states encompass approximately 739,769 km^2 and contain portions of both the Missouri and Mississippi Rivers. There are 632,740 kilometers of stream mapped at a scale of 1:100,000 within EPA Region 7 with 153,335 kilometers of permanent flowing water. Some of the other rivers in Region 7 exhibit very braided channels. The Platte River is a good example of a major river that is extensively braided. EPA Region 7 comprises portions of 16 of Bailey's Ecological Sections. These ecoregions range from the Ozark Highlands to the South Central Great Plains to the Nebraska Sand Hills (Figure 2).

Much of the region outside of the Ozark Highlands is dominated by agricultural and grazing activities. These two land uses and affiliated features like ditches and water diversions probably represent the most widespread threats across the region. There are numerous large impoundments in the region including Lake of the Ozarks and Table Rock Lake. Mining and extractive activities are common in the region as well. Oil and gas wells are prominent in

Kansas and western Nebraska, coal mines are widespread in Missouri and Iowa, and lead mines in Missouri. Although not heavily populated by eastern United States standards, Missouri has the highest population density of the states comprising EPA Region 7. Maps of the threat datasets described here and others used for the project are available in Appendix B. Certainly each state in Region 7 has at least some of nearly every threat quantified as part of this project. Although it is not possible to discuss all of the threats in each state we will highlight some of the more apparent distinctions between the four EPA Region 7 states.

Iowa is a state dominated by row crop agriculture and many of the threats in Iowa like crop chemical use and ditched or diverted streams are related to agricultural activities. Outside of agriculture, the main threat distinction in Iowa is the presence of coal mines. Kansas is also dominated by agricultural activities. Other threats that stand out in Kansas are the presence of dams and numerous oil and gas wells. Missouri has a higher population that any of the other states in EPA Region 7. Channelized or ditched streams are prolific in the Missouri Alluvial Basin of Missouri as well as in the Central Dissected Till Plains in northern Missouri. Along with Iowa, Missouri has many coal mines. One primary distinction in Missouri is the presence of numerous lead mines primarily within the Ozark Highlands. Nebraska has many agricultural activities in the south and eastern portions of the state in conjunction with many channelized or ditched streams in these same areas. Oil and gas wells are prominent in southwestern Nebraska.

Figure 1. EPA Region 7.

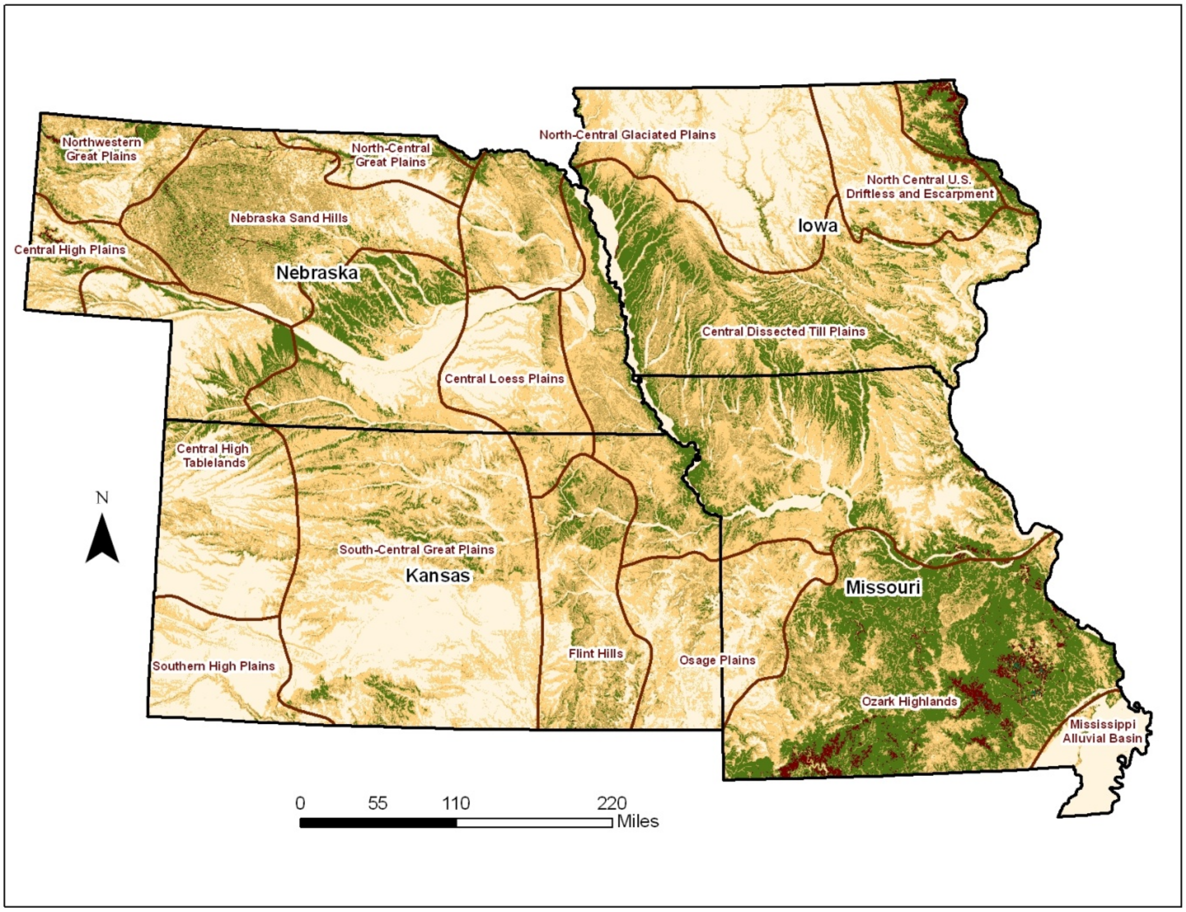


Figure 2. Landforms and ecoregions of EPA Region 7 (C. Diane True map).

Chapter 2

Threat Data, Assessment Units, and Quantification

There are a multitude of threats or stressors that negatively affect the ecological integrity of riverine ecosystems (Allan and Flecker 1993; Richter et al. 1997). The first step in any effort to account for anthropogenic stressors is developing a list of candidate causes (U.S. EPA 2000). We sought to utilize as many different threat data sets as possible. A caveat was that we wanted the data to be consistent and useful across state lines so that the entire drainage area could be considered for streams beginning in one state and flowing into another. To help identify all possible threats we relied on our regional oversight committee to help us generate a list of possible threats. From this list we identified those that were available digitally and those that we could create within the time and money constraints for the project (Figure 3). Table 1 presents a list of all threats that were quantified as part of this project including sources and dates. Each of these individual threat datasets has its own limitations with regard to data quality, completeness, resolution, and date of mapping. It should be noted that some threats are represented by multiple components; for instance roads are represented as both 1) length of road and 2) road-stream crossings. Many of the threats that were mapped as points had distance-to-threat information computed.

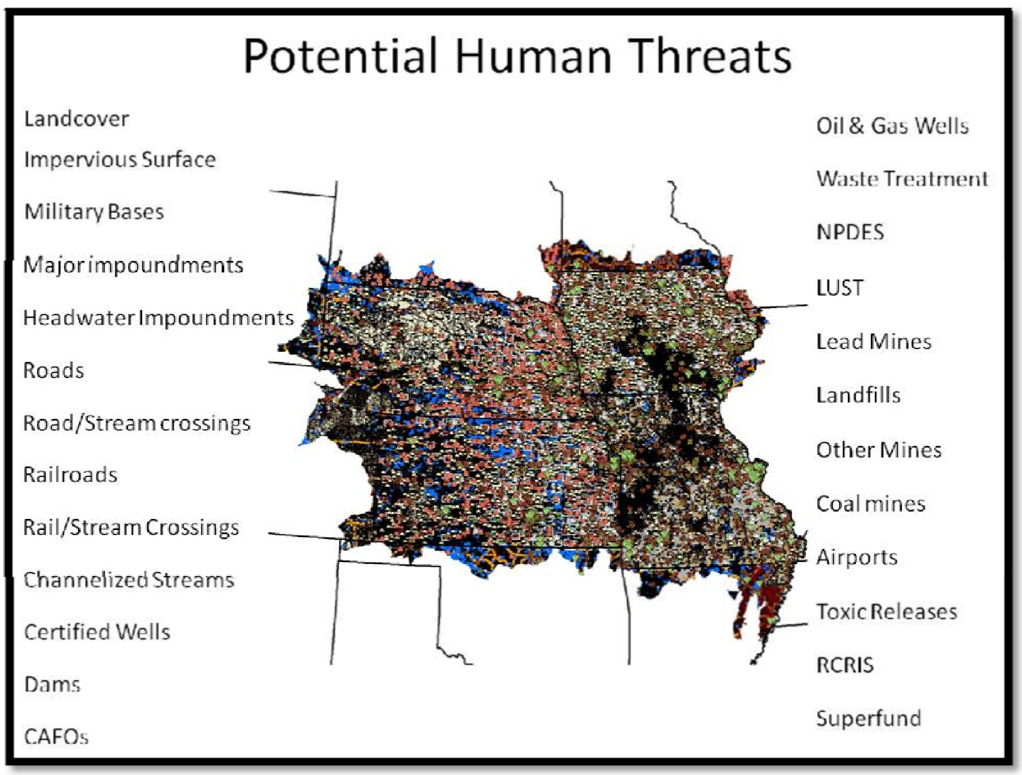


Figure 3. Selected potential human threats to aquatic ecological integrity.

A substantial portion of project time was spent looking for, acquiring, and cleaning geospatial data representing these anthropogenic threats to aquatic ecosystems. It is important to note that many of the data layers acquired for this project had very minimalistic metadata available. Appendix B contains brief reports for each input data set utilized for this project.

Table 1. The primary threat data layers utilized for this project with source and data set date.

File Name	File Definition	Source(s)	Source Date
Airports.shp	Airports	Acquired from EPA. EPA cites: GDT Dynamap 2000	2000
CAFOS.shp	Confined Animal Feeding Operation	Acquired from EPA. EPA cites: Dunn & Bradstreet 2003	2003
Mines.shp	Mines, except coal and lead	USGS	2005
Coal_Mines.shp	Coal mines	EPA BASINS	2001
		University of Nebraska - Lincoln	1996
		Iowa DNR	2003
Lead_Mines.shp	Lead mines	EPA BASINS	2001
Oil_Gas_Wells.shp	Active oil and gas wells	Conservation and Survey Division, University of Nebraska – Lincoln	1996
		Kansas Geological Survey	Varies
		MoDNR	Provisional
IA_wells.shp	Iowa certified water wells	Iowa DNR	1995
KS_wells.shp	Kansas certified water wells	Kansas Geological Survey	Varies
MO_wells.shp	Missouri certified water wells	MoDNR	2006
NE_wells.shp	Nebraska certified water wells	Nebraska DNR	2006
Roads.shp	Roads	TIGER census	1999
Road_Stream_Crossings.shp	Road and stream intersections	MoRAP	2007
Railroads.shp	Railroads	TIGER census	1999
Rail_Stream_Crossings.shp	Railroad and stream intersections	MoRAP	2007
RCRIS.shp	Resource conservation recovery information system sites	EPA	2007
Superfund.shp	Superfund sites	EPA	2007
Toxic_Releases.shp	Toxic release sites	EPA	2007
NPDES.shp	National pollutant discharge elimination system sites	EPA	Unknown Acquired 2007
WWTF.shp	Waste water treatment facilities	EPA	Unknown Acquired 2007
Landfills.shp	Landfills	EPA BASINS	2001
LUST.shp	Leaking underground storage tanks	MoDNR	2004
EPA_R7_Channelized_Streams.shp	Channelized or ditched streams	Kansas Dept of Health and Environment	unknown
		Iowa DNR	2003
		Nebraska DEQ	Unknown
		Created by MoRAP using: 24K NHD, 100K NHD, and NWI	2008
Major_Impoundments.shp	Major impoundments	Created by MoRAP using: NHD and NWI	2008
EPA_R7_Headwater_Impoundments.shp	Headwater impoundments	Created by MoRAP using: NWI and 2001 NLCD	2008
Dams.shp	Dams	U.S. Army Corps of Engineers	1996
Military_Bases.shp	Military bases	Bureau of Transportation	2001
Crop_Pest (ESRI grid)	Estimated crop pesticide application	Created by MoRAP using various inputs including 2001 NLCD	2007
Impervious (ESRI grid)	Impervious surface areas	Created by MoRAP using: 2001 NLCD	2006
NLCD_2001 (ESRI grid)	2001 national landcover dataset	USGS/MRLC	2006
1990 Population (geodatabase)	1990 block population	Census Bureau	1990
2000 Population (geodatabase)	2000 block population	Census Bureau	2000

Special Data Sets Created for this Project

Although the intent for this project was to use existing geospatial data sets of potential threats to aquatic ecosystem integrity, our regional oversight committee identified several threats that were deemed particularly important, but for which no existing data set was available. On advice from the regional oversight committee we undertook to create these data sets. These data sets include:

1. Fragmentation of stream networks from major impoundments
2. Headwater impoundments
3. Ditched or channelized streams
4. Population change in the drainage area above every stream segment
5. Estimated crop pesticide use (MoRAP recreated an existing data set at a finer resolution using newer data)
6. Modified impervious surface from 2001 NLCD (MoRAP altered the impervious surface from the 2001 NLCD)

Stream Fragmentation

A data set that deserves special mention is one representing stream fragmentation that was developed specifically for this project. We sought to determine how fragmented the stream networks were from impoundments. We wanted to answer questions like; What is the total length of interconnected stream in any given "network fragment"? In other words, how many miles of stream does a fish have access to without having to swim through an impoundment?

Any stream barrier intersecting Small or Large Rivers was used to cut up the stream network. Any segments intersecting or inundated by an impoundment were given a distinct code to temporarily remove them from the network. All remaining interconnecting stream segments were given a "Group_id" and their total length was summed. This information yields the total length of all stream network in any given network fragment (Figure 4).

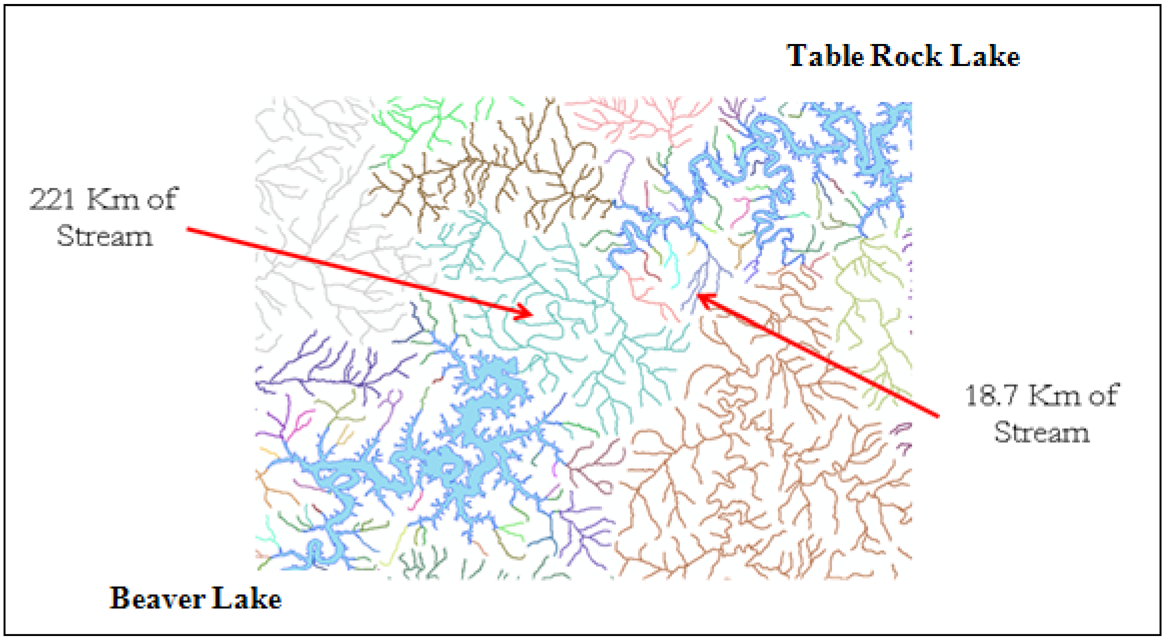


Figure 4. Representation of the length of interconnected stream by stream fragment. In this example the stream networks are fragmented by Table Rock and Beaver Lakes. Thought of a different way, a fish living in the green fragment between lakes would have access to 221 kilometers of stream without having to swim through impounded water, while a fish living in the smaller purple headwater fragment would have access to only 18.7 kilometers of stream without having to swim through an impoundment.

Headwater Impoundments

Headwater impoundments were identified as an important threat for this project. We wanted to determine an approximate number of headwater impoundments in EPA Region 7. First we extracted waterbodies from the National Wetlands Inventory (NWI) and National Land Cover Dataset (NLCD) and combined into one layer. Then we used a DEM to create a very dense stream network representing small headwater streams. Any waterbodies that intersected these small headwater streams were extracted for analysis. We then compared the result to various other sources to ensure we only had manmade headwater impoundments. See Appendix B or the headwater impoundment layer's metadata for more information.

Channelized Streams

We attributed our base stream layer (1:100,000 modified NHD) with an attribute indicating whether each stream segment was channelized or ditched. This was accomplished by creating a new layer of channelized or ditched stream segments using attribution from the 1:24,000 NHD, attribution from the NWI, and manual assessment of stream segments. This new layer of ditched stream segments was used to attribute our base assessment units (1:100,000 modified NHD). The resulting attribution approximates channelized or ditched stream segments in the 1:100,000 modified NHD. See Appendix B or the channelized stream layer's metadata for more information.

Population Change

To quantify population change in the watershed above every stream segment we used the 1990 and 2000 census block information for EPA Region 7 and assigned the population from each census block to our assessment catchment polygons based on the percentage of block area located within each catchment. This process yielded a population from 1990 and 2000 attached to each catchment polygon. Once complete we were able to quantify the population from each census year for the drainage area above each stream segment. Subtracting these two values resulted in population change. It should be noted that although we computed population change, our human threat index utilized population data for the year 2000. See Appendix B for more information.

Estimated Crop Pesticide Use

We created a grid of estimated crop pesticide use using methods developed by the United States Geological Survey (USGS) (Gianessi and Thelin 2000; Nakagaki 2007). We used the 1997 agriculture census data's pesticide sales by county for the 43 most used crop pesticides. The amount of pesticide sold by county was partitioned evenly to pixels of cropland in a given county using the 2001 NLCD. The resulting grid displays the approximated amount of pesticide used on each 30 meter grid cell. See Appendix B or the crop pesticide grid's metadata for more information.

Modified Impervious Surface

We modified the "developed" classifications from the 2001 NLCD by removing most rural roads from the impervious class. This was done because a 30 meter pixel is often too large to represent most rural roads and overestimates impervious surface in these areas (Figure 5).
A shrink and expand process was used to remove rural roads, but maintain urban impervious. See Appendix B or the impervious surface grid's metadata for more information.

Figure 5. Figure depicting impervious surface grid cells (red pixels) draped over a National Agricultural Imagery Program (NAIP) image. This figure highlights the overestimation of impervious surface from the 2001 NLCD.

Assessment Units

The primary assessment units consist of catchment polygons for all of the primary channel stream segments from a modified version of the 1:100,000 National Hydrography Dataset (NHD) (Sowa et al. 2007). These catchment polygons were created by using a 30-meter digital elevation model (DEM) and the stream network (Figure 6). These input data sets were put into an automated process in ArcMap to create the catchment polygons. Although quite variable, the average size of a catchment polygon assessment unit is 2-3 square kilometers. This small size allows for very fine assessments. The resulting polygons carve EPA Region 7 into approximately 400,000 individual hydrologic pieces.

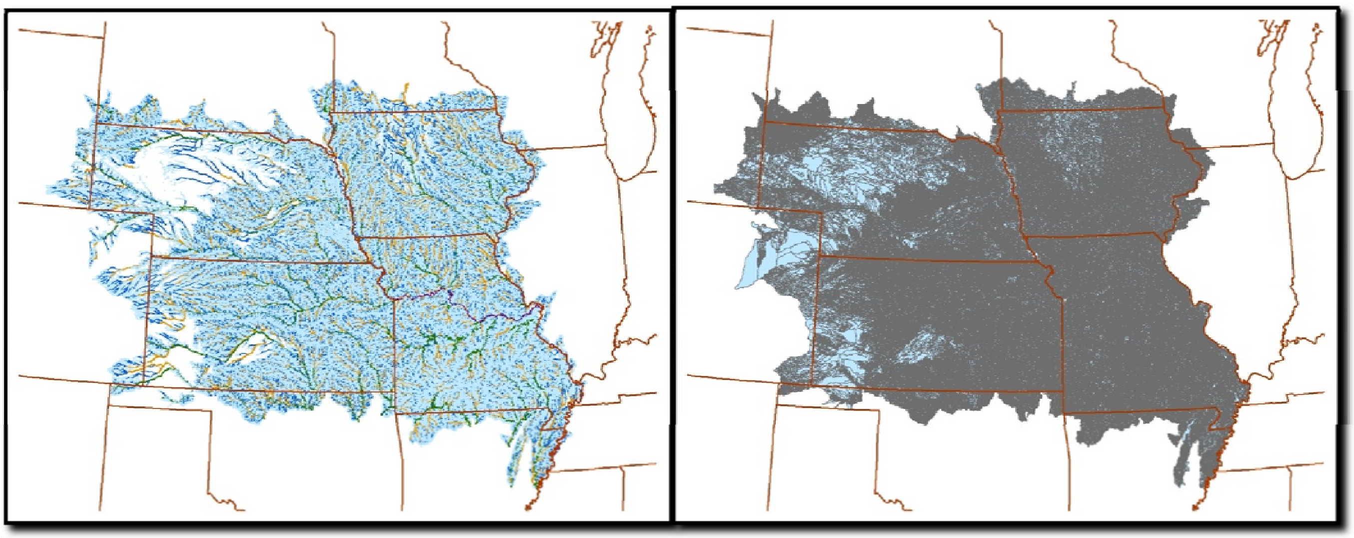


Figure 6. Approximately 400,000 primary channel stream segments (left) and corresponding catchment polygons (right).

The utility of the catchment polygons resides in the fact that there is a one-to-one relationship between the catchment polygons and the stream segments (Figure 7). This allowed the transfer of data from the polygons to the stream networks and downstream accumulations to be computed. The common identifier "Seg_id" allows table relations to be performed between the two files. Statistics such as total drainage area, point sources, landcover, etc. can then be attributed to the streams and their values can be converted to a proportion of the drainage area.

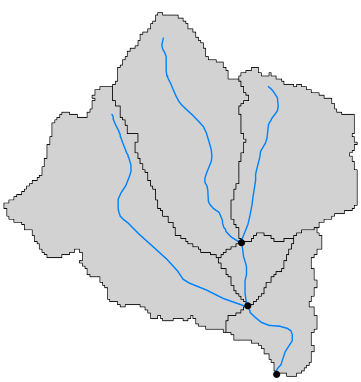
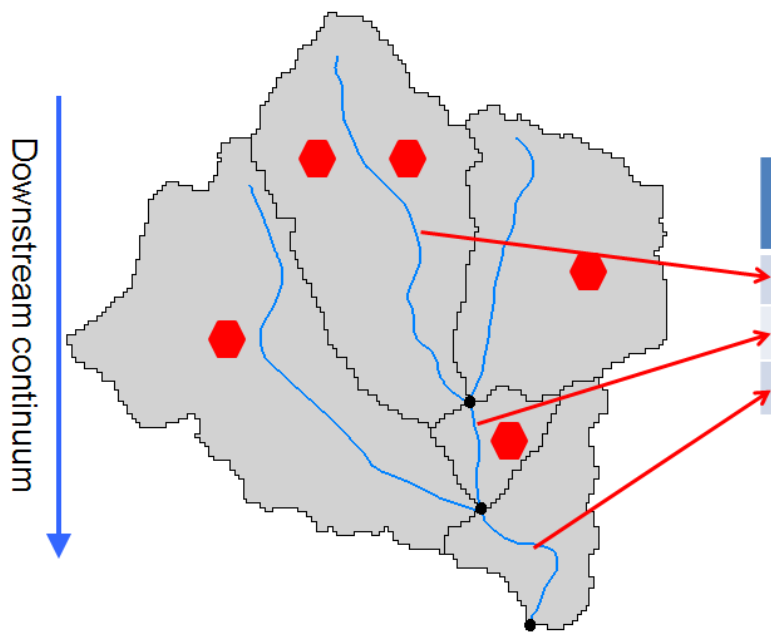


Figure 7. Zoom-in of five catchment polygons illustrating one catchment polygon for each stream segment (blue lines).

A secondary assessment units layer consists of stream buffers for each of the primary channel stream segments in the modified 1:100,000 NHD. Headwaters and Creek stream size classes were buffered by 45 meters on a side, while Small and Large Rivers were buffered by 110 meters on a side. The Missouri and Mississippi Rivers were buffered by 110 meters from the stream bank. These buffers were used to quantify riparian land cover.

Quantifying the Data

Each individual threat data layer was quantified by first tabulating locally to get an amount in each catchment polygon. The next step involved bringing the local information over to the stream network. Then programs were run to quantify everything in the drainage area above every stream segment. This basic idea is depicted in Figure 8. The same basic process was used to quantify continuous data (land cover), point data (CAFOs), and linear data (roads).



Toxic Releases

# Local	# In Total Drainage Area
2	2
1	4
0	5

Figure 8. Example depicting the quantification of point sources upstream of each stream segment. Blue lines represent stream segments and grey polygons represent the catchment polygons.

The quantified data consists of local amount and the amount in the total drainage area for each individual stream segment. Finally, depending on the type of data that was quantified, there are fields that present the data as amount per unit area for the local and total watershed (percent of the watershed, number per square kilometer, or length per square kilometer). All of this information resides in tables that relate to the stream and catchment layers via the identifier “Seg_id”. Figures 9 and 10 depict CAFOs and cropland quantified for each stream segment’s drainage area.

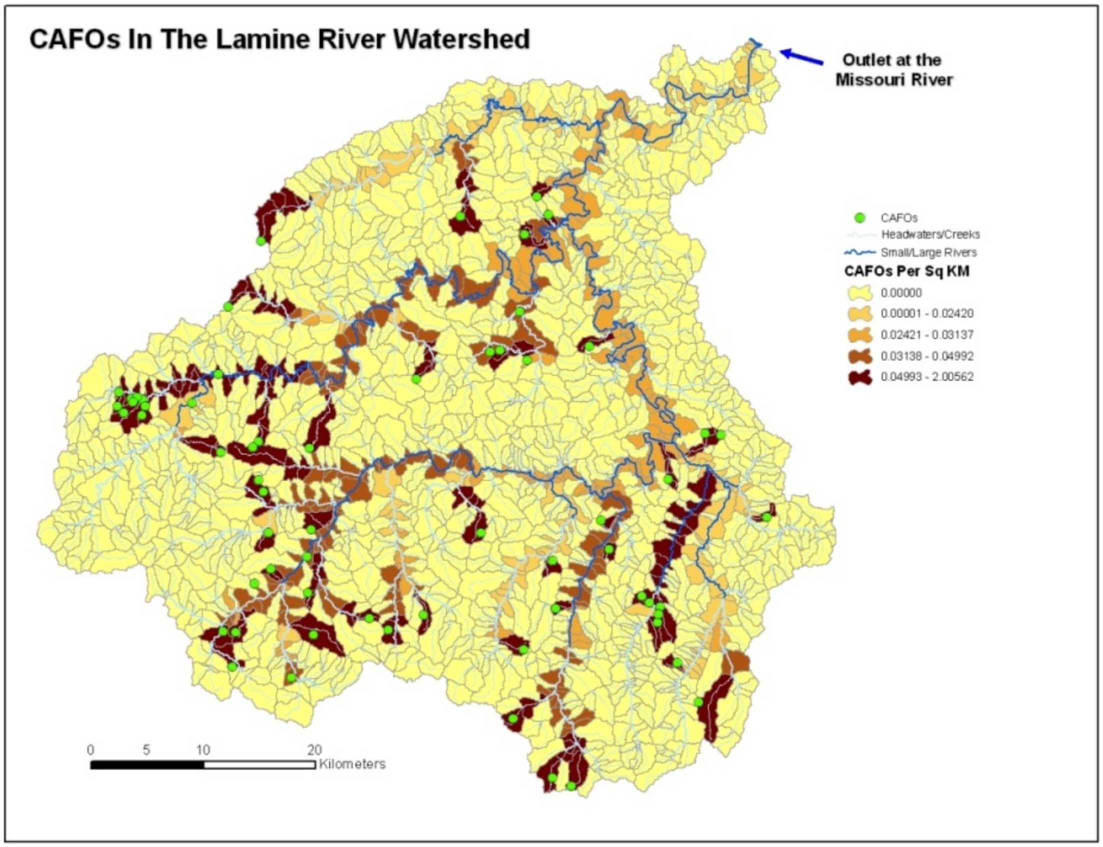


Figure 9. Number of CAFOs per square kilometer in the drainage area above <u>each</u> stream segment within the Lamine River watershed.

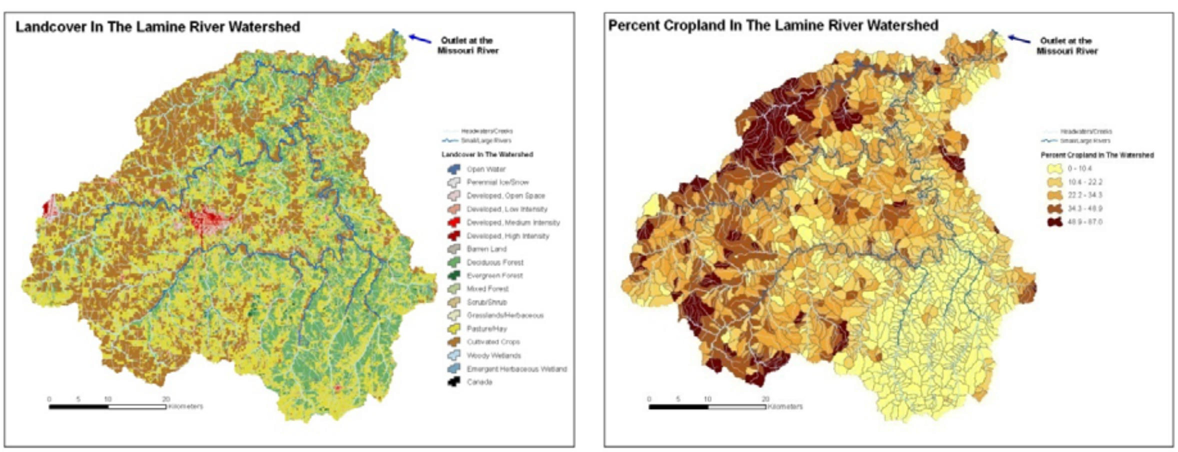


Figure 10. Land cover from the NLCD in the Lamine River Watershed (left map) and percent of drainage area in cropland above <u>each</u> stream segment (right map).

Beyond simply quantifying how much or how many of a given threat is upstream we also wanted to consider the spatial distribution when possible (Figure 11). Because nearby threats have a different potential impact on ecological integrity than the same threat further away we computed both minimum and mean distance to threat for threats represented as points on the landscape. Minimum distance to a threat is represented as the in-stream distance to the closest threat upstream. Mean distance is represented as the mean in-stream distance to all threats of a given class (e.g. CAFOs, coal mines, and others) upstream (Figure 12).

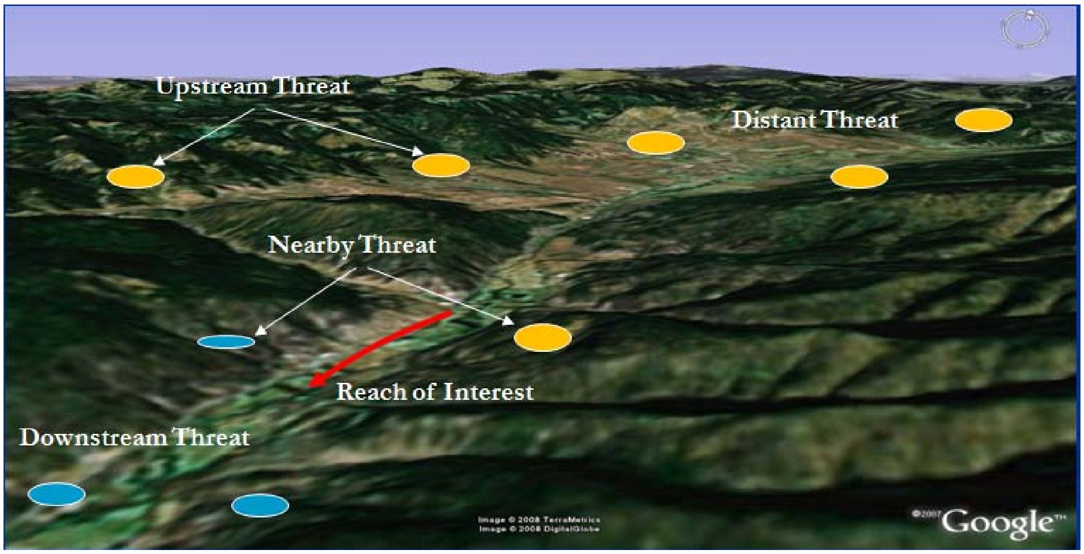
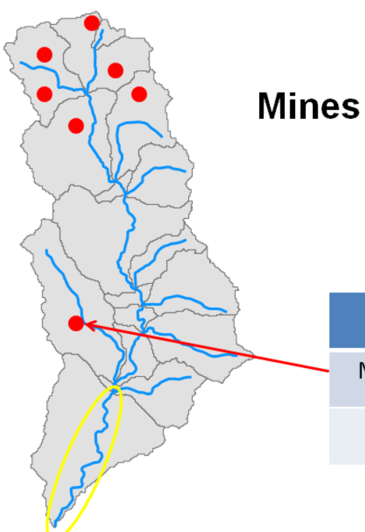


Figure 11. Ecological integrity of riverine ecosystems is dependent on the integrity of the entire watershed.



Mines Upstream	
Minimum Distance	3 Km
Mean Distance	16.5 Km

Figure 12. Figure depicting both minimum and mean distance to mines in a watershed. Both of these pieces of information have been quantified and are available in the data package.

Data Gaps and Limitations

It is important to address the limitations of the data compiled and quantified for this part of the project. The data quantified as part of this project in no way represents every possible threat to aquatic ecosystem integrity. In reality it is a reflection of data that were identified as being a potential threat and, as importantly, data that were available for use across a large geographic area. All data utilized represents the best available source data at the time of the project given the constraints of requiring data that is seamless across state borders.

The metrics quantified as part of this project relied on numerous existing datasets each with its own inherent limitations and inaccuracies. Each individual input data set has its own date of creation, resolution, standards, and level of completeness. Point data sets tend to imply an absolute location on the landscape, however, we learned that this is very often not the case.

There were three basic issues encountered with source data; 1) location or horizontal positioning on the landscape, 2) incompleteness, and 3) having multiple sources of the "same" data. As mentioned previously, many of the datasets depicting features on the landscape using points suffered from poor point positioning. An example of this is the National Pollution Discharge Elimination System (NPDES) layer acquired from the Environmental Protection Agency (EPA). Ideally each point in this data layer would be located at the "end of pipe", however features are located by a variety of means including but not limited to facility address match, nearest intersection, owner's address, map interpolation, or centroid of census block. Figure 13 gives an example of CAFOs taken from the NPDES data layer that represent owner address as opposed to facility location.

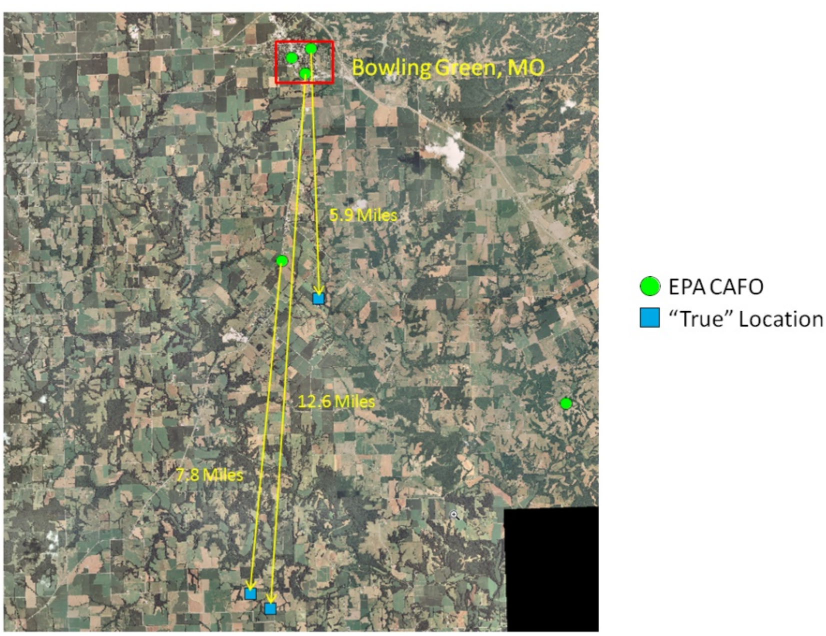


Figure 13. Figure depicting facilities mapped by owner's address as opposed to facility location.

As might be expected, most data sets had some limitations with regard to incompleteness resulting from file date or mapping protocol where all features from ground reality are not represented in the data layer.

The final data issue we encountered was with regard to having several data sets representing the same threat, but with clear differences (Figure 14). When this occurred we talked to the originators of each data set when possible and made our best professional judgment on which data layer was best to use.

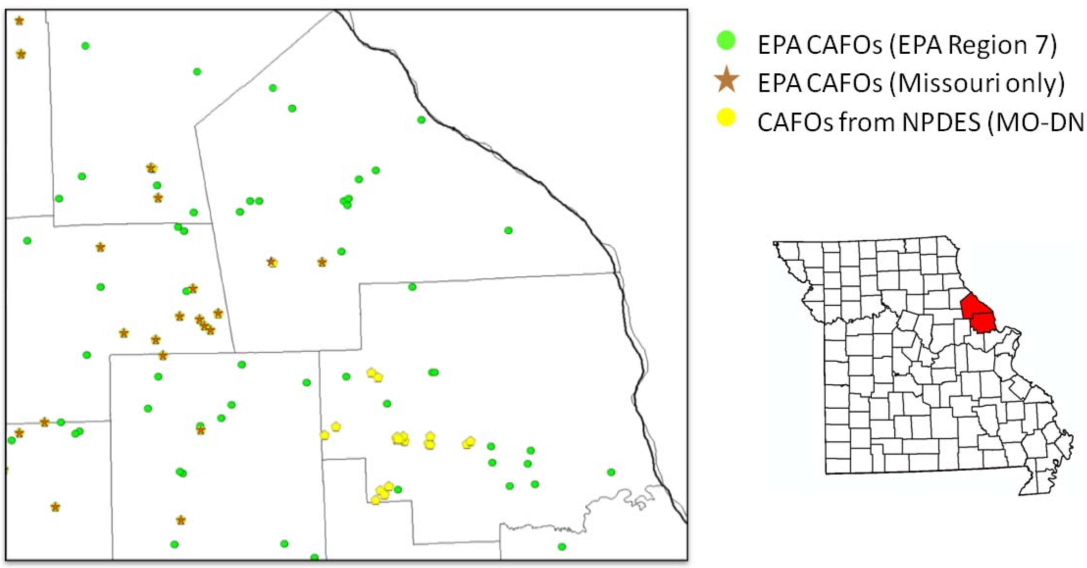


Figure 14. Three data layers representing confined animal feeding operations (CAFOs) in Missouri.

Chapter 3

Human Threat Index (HTI) Creation and Validation

Human Threat Index (HTI) Creation

A very common approach for measuring human disturbance on streams is through the use of multimetric biological indicators like indices of biotic integrity (IBI) for both macroinvertebrates and fish (Rosenberg and Resh 1993; Rabeni et al. 1997; Karr and Chu 1999). One of the shortcomings of biological indicators like IBIs is that these indices can only be developed for locations where biological data has been collected, which is often a very small percentage of the stream miles within any study area. In addition, these indices often do not provide information as to the direct cause for diminished ecological integrity (USEPA 2000; Wang et al. 2008). Creating an index directly from anthropogenic threats on the landscape that can be validated using biological indices would provide a “seamless” depiction of ecological integrity throughout a stream network and could be used to ascertain specific causes of impairment. This “seamless” depiction of ecological integrity throughout EPA Region 7 is what this project attempted to accomplish.

As stated previously we sought to create a human threat index (HTI) that utilized as many potential threats to aquatic ecological integrity as possible. We ultimately combined 35 individual threats that were quantified for the local catchment, the upstream contributing area, and riparian buffer into an overall HTI. Table 2 lists the data that went into the HTI.

Table 2. The individual threats that were ultimately quantified and incorporated into the human threat index (HTI). The 15 threats with an asterisk * were distance weighted by both minimum distance and mean distance.

1.	Impervious surface	19.	Railroad and stream intersections
2.	Cropland	20.	Waste water treatment facilities *****
3.	Pasture	21.	Toxic release inventory sites *
4.	Riparian Impervious surface	22.	Resource conservation recovery information system *
5.	Riparian Cropland	23.	Estimated kilograms of crop pesticide
6.	Riparian Pasture	24.	Length of railroads
7.	Airports *	25.	Pipelines
8.	Military bases *	26.	Landfills *
9.	Lead mines *****	27.	Headwater impoundments
10.	Coal mines *****	28.	Confined animal feeding operations *
11.	Dams *****	29.	Dollar amount of livestock sales
12.	Road and stream intersections	30.	National pollution discharge elimination system sites *
13.	Certified water wells	31.	Length of channelized or ditched streams
14.	Superfund sites *****	32.	Leaking underground storage tanks *
15.	Major impoundments	33.	Population from the 2000 census
16.	Length of roads	34.	Distance to lakes (Local values only)
17.	Oil and gas wells *****	35.	Stream fragmentation (Local values only)
18.	Mines excluding coal and lead mines *****		

Distance Weighting

As described in the previous section, the distance to an individual threat of a given type impacts the potential threat to a given stream segment. For example, a coal mine that is situated just a few kilometers upstream has a different potential impact than the same coal mine situated 100 kilometers upstream. In addition, two watersheds with the same density of a given threat may have very different spatial arrangements of those threats and therefore different impacts from those same threats. For instance, in a given watershed ten coal mines located immediately upstream of the outlet would have a different potential threat to the outlet stream segment than would ten coal mines located 200 miles upstream even though the density of coal mines in these two scenarios would be identical.

To account for these different spatial arrangements of threats we applied both minimum distance and mean distance weightings (Figure 15). The same weighting multiplier was applied to minimum and mean distance with higher weights applied to threats nearby (Table 3). Table 4 presents an example of how these distance weights were applied to four different watersheds each with the same density of mines. Even though each of these watersheds had the same density of mines the spatial arrangement was quite different. As a result they each received a very different distance weighted score. Although we would have liked to distance weight each of the 35 threats incorporated into our HTI for technical reasons associated with data types and the physical number of points in some datasets we were only able to distance weight 15 threats.

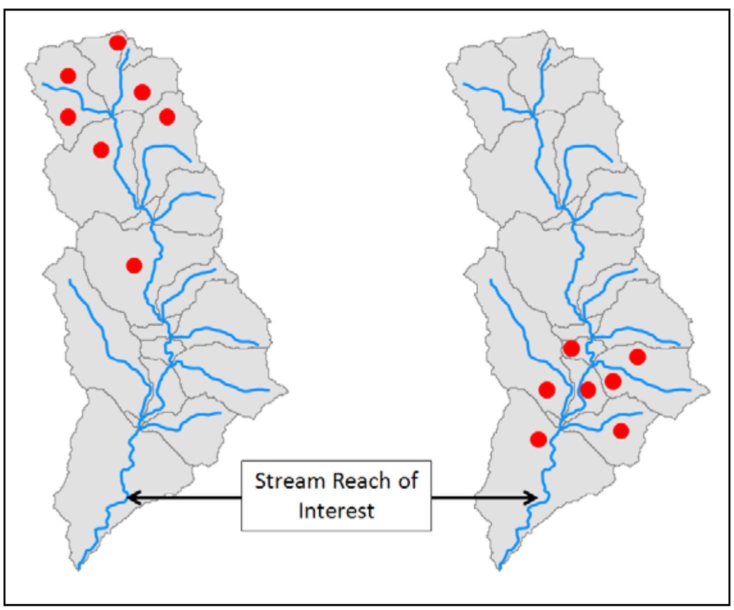


Figure 15. Example of two hypothetical watersheds of the same size with the same number and density of coal mines (red dots), but with very different spatial arrangements in relation to the stream reach of interest. In this example the stream reach of interest in the map on the right probably has more potential threat/impact from the mines than does the same segment in the map on the left.

Table 3. Weights (multipliers) applied to threats based on minimum and mean distance classes.

Weight (multiplier)	Minimum Distance (Km)	Mean Distance (Km)
15	0 - 2	0 - 2
7	2 - 10	2 - 10
3	10 - 100	10 - 100
1	> 100	> 100
NA	None	None

Table 4. Example of four different watersheds (tables rows) each with the same density of mines (0.03 per km^2), but with different spatial arrangements and therefore distance weighting. The density of mines was multiplied by the minimum distance weight and then by the mean distance weight to yield a distance weighted score. Larger numbers indicate more potential threat.

Mine Density ($\#/Km^2$)	Minimum Distance Weight	Mean Distance Weight	Distance Weighted Score
0.03	3	1	0.09
0.03	7	3	0.63
0.03	15	3	1.35
0.03	15	15	6.75

Variable Contribution Weighting Considerations

Early in the project we conducted a survey that was made available over the internet to members of our regional oversight committee and other “experts” with the intent of establishing weights for our individual threats in relation to each other. This survey consisted of a list of potential threats that had been generated at our first two regional oversight committee meetings. Those surveyed were asked to rate the perceived impact of each threat to five components of ecological integrity (physical habitat, water quality, flow regime, energy/nutrient dynamics, and biotic interactions). Participants used a rating scale of: 0=No Impact; 1=Low Impact; 2=Moderate Impact; 3=High Impact. We received 24 completed surveys. The results of these surveys were used to compute a Mean and Mode from the responses for each threat. However, because of potential biases from a limited number of responses and varying interpretations of the threats and potential impacts from the threats we elected not to utilize this survey for weighting. The results of this survey are presented in Appendix D.

Although there are certainly differences between specific threats and their impacts on stream ecological integrity, establishing relationships and thresholds between differing threats is difficult. Part of the difficulty lies in the fact that most available data does not provide dose information (i.e. what and how much a given threat discharges at a given site). Another difficulty lies with determining how many/much of a given threat has the same ecological response as another threat (e.g. how many road/stream crossings does it take to have the same ecological impact as 2 kilometers of ditched stream?).

Ranking and Rescaling Individual Threats

Because the varying input threat datasets were quantified with different units of measure (i.e. percent area, point densities, distance weighted scores, length per unit area, and other) we needed to standardize these units before creating our threat index. We elected to rank and rescale the quantified threats relative to two stream size groupings. This was done so that headwaters and creeks would be compared to other similarly sized streams and not compared to larger streams that invariably have more cumulative threats.

Using SPSS Version 14, threats quantified at the local scale were ranked from 1 to n for each stream segment within two stream size groupings; 1) Headwaters/Creeks and 2) Small Rivers and larger. Within each of these groupings the stream segment with the lowest density or amount of a given individual threat received a rank of 1, while the segment with the highest density received the highest rank. Segments with the same density received the same rank. After the ranking was completed the results for each individual threat were rescaled from zero to 100 with 100 representing the highest potential threat for an individual variable. This process was done for each individual threat at the local scale. The resulting metrics for each local threat were then summed to generate a raw local human threat value. Raw local threat index values were subsequently rescaled from zero to 100. This index is referred to as the local human threat index (LHTI) (Figure 16). LHTI values approaching 100 represent the most threatened sites based on local conditions.

This same process was repeated for individual threats quantified at the watershed scale (contributing area) for each stream segment. The individual watershed threats were summed and rescaled from zero to 100 to create a watershed human threat index (WHTI) (Figure 17). The LHTI and WHTI were subsequently added together with the result rescaled from zero to 100. This process assumed equal influence between the summed threat categories. The combined index represents the overall human threat index (HTI) for each stream segment throughout EPA Region 7 (Figure 18).

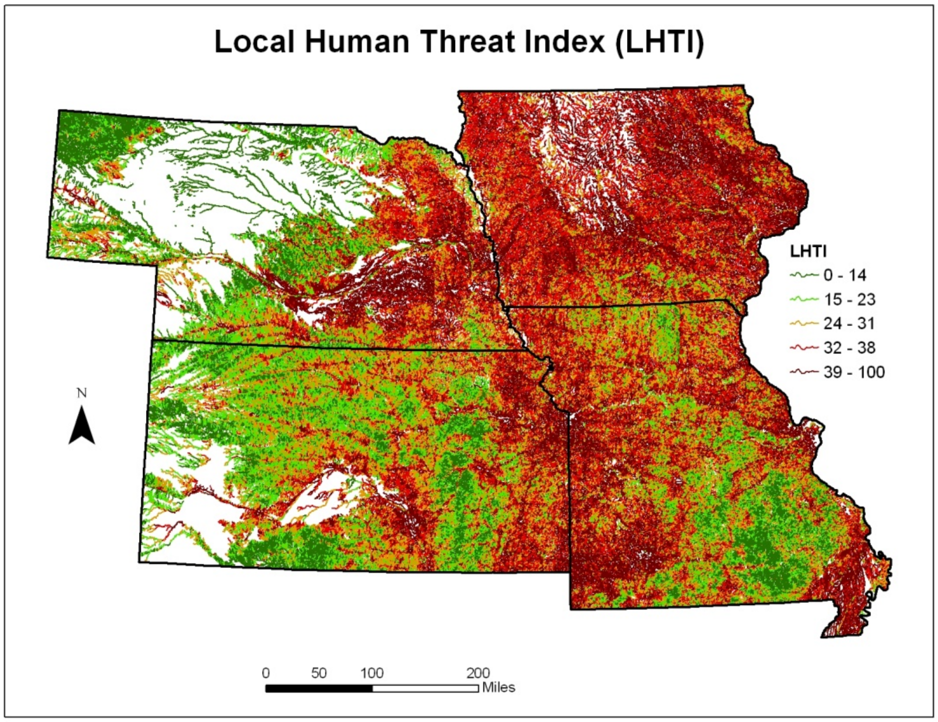


Figure 16. Local human threat index (LHTI).

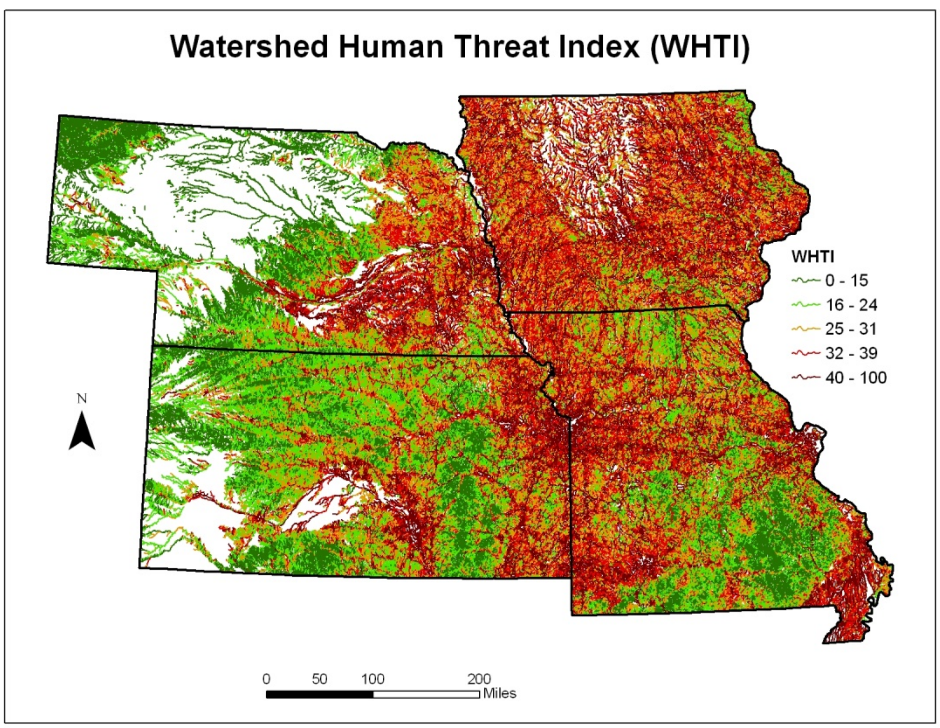


Figure 17. Watershed human threat index (WHTI).

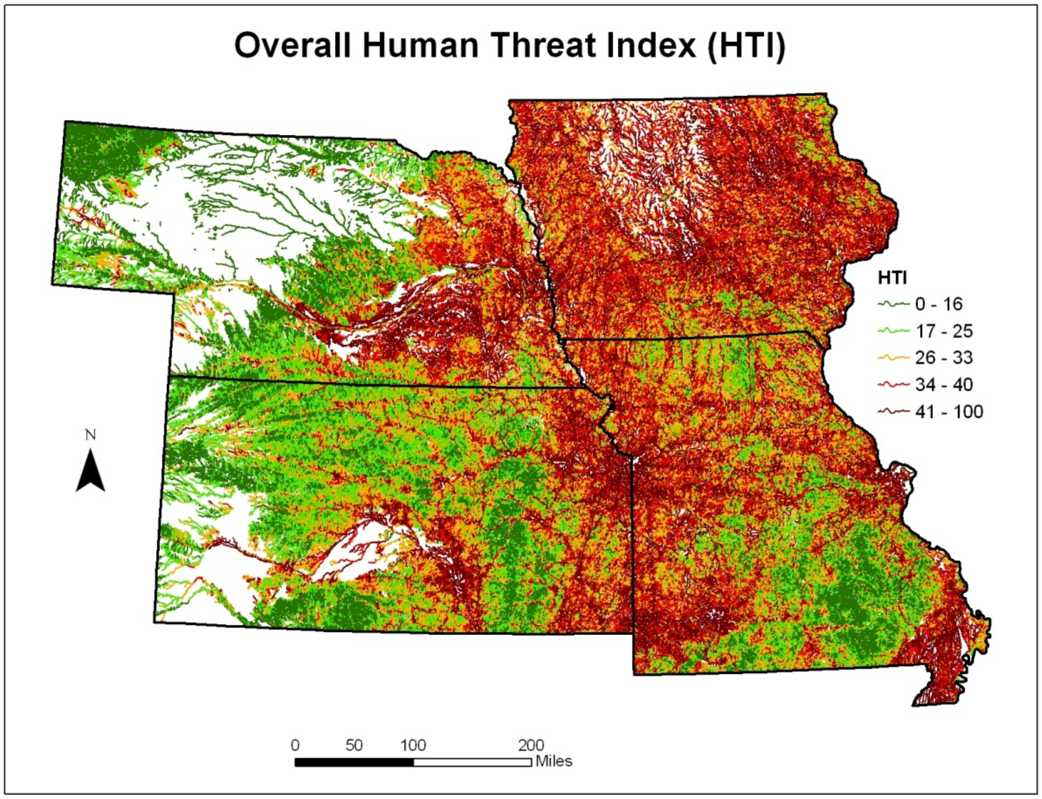


Figure 18. Overall human threat index (HTI).

Human Threat Index (HTI) Validation

As described previously a common approach for measuring human disturbance on streams is through the use of multimetric biological indicators like IBIs for both macroinvertebrates and fish. These same biological indicators can also be used to help verify or validate an index that combines quantified anthropogenic features on the landscape into an overall threat index such as the HTI created for this project.

Because the development of any index of potential human threat to aquatic ecological integrity is somewhat subjective we wanted to test whether the index was a good predictor of in-stream ecological condition. To see how well our human threat index (HTI) accounts for ecological integrity we tested our HTI against both macroinvertebrate indices and a fish Index of Biological Integrity (IBI). The results of these analyses generally indicate that our overall HTI is a good predictor of macroinvertebrate integrity, but is less useful for predicting fish IBIs.

We initially planned on validating our HTI using fish and macroinvertebrate collections from nine locations in Missouri that the Missouri Department of Conservation sampled for us specifically as part of this project (Figure 19). These nine collection locations were selected within two of the Ecological Drainage Units where Resource Assessment and Monitoring (RAM) field crews were sampling during the summer of 2008. Several site options were provided to the field crew so that three collections would be made from low, medium, and high threat sites based on a preliminary HTI. Both fish and macroinvertebrate data collected from these nine sites was submitted to EPA's STORET database. Although this data was utilized we decided to include other fish and macroinvertebrate data from sites across Missouri and EPA Region 7. This substantially increased the number of observations in our validation set.

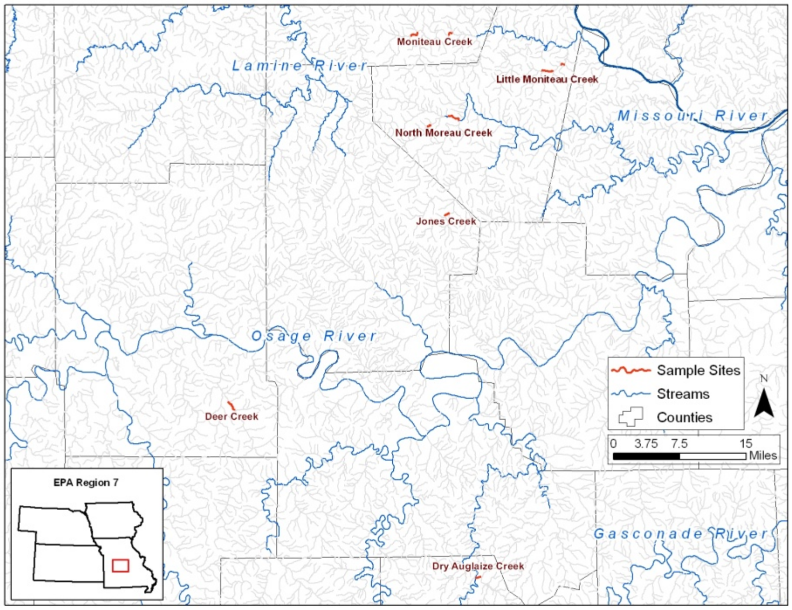


Figure 19. Sites sampled by the Missouri Department of Conservation in August 2008 to help validate the human threat index. Fish and macroinvertebrate indices were developed for these locations using Resource Assessment and Monitoring (RAM) protocols.

HTI Comparison to Macroinvertebrate Metrics/Indices

Benthic macroinvertebrates are frequently used as a bio-monitoring measure for assessing the ecological condition of water bodies (Karr and Chu 1999; Whiles et al. 2000) and are generally considered a good indicator of the physical, hydrologic, and chemical conditions of streams (Rosenberg and Resh 1993; Wang and Kanehl 2003; Wang and Lyons 2003). Macroinvertebrates are probably the most often used organism to assess water quality (Rosenberg and Resh 1993).

Although there are various metrics or indices of macroinvertebrate community condition research conducted by Barbour et al. (1992) and Rabeni et al. (1997) in Missouri concluded that Taxa Richness (TR), Ephemeroptera/Plecoptera/Trichoptera Taxa Index (EPTT), Biotic Index (BI), and Shannon Diversity Index (SDI) are reliable and sensitive macroinvertebrate indices (Sarver et al. 2002).

The Missouri Department of Conservation began the Resource Assessment and Monitoring (RAM) Program in 2000. The RAM program looks at both fish and macroinvertebrate communities to determine indices of biotic integrity for warm water wadeable streams. We acquired RAM macroinvertebrate data from the Missouri Department of Conservation which included TR, EPTT, BI, and SDI metrics. The dates of collection ranged from 2002-2009. Collections used the protocols described in Sarver 2001. These data were joined to our stream segments in a GIS which yielded 415 locations with computed macroinvertebrate metrics which included the nine sites sampled specifically for this project (Figure 20). Integrating these data with our HTI data sets allowed us to compare our HTI to each of the four macroinvertebrate indices.

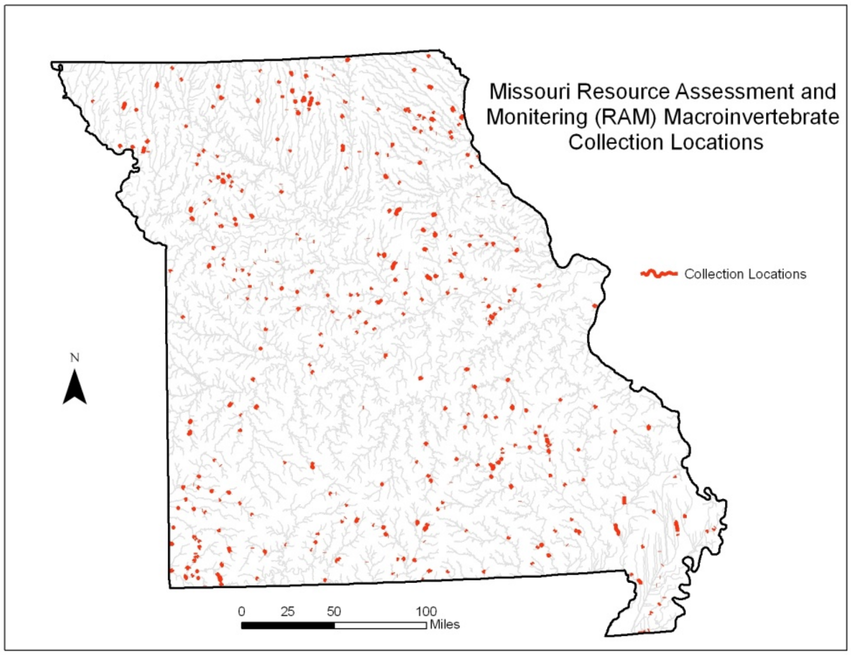


Figure 20. Missouri Resource Assessment and Monitoring (RAM) macroinvertebrate collection locations (415 locations).

Sarver et al. 2002 (*Biological Criteria for Wadeable/Perennial Streams of Missouri*) defines the four macroinvertebrate metrics as follows:

> Taxa Richness reflects the health of the community through a measurement of the number of taxa present. In general, the total number of taxa increases with improving water quality, habitat diversity, and/or habitat suitability.
>
> Ephemeroptera/Plecoptera/Trichoptera Index (EPTT) is the total number of distinct taxa within the orders Ephemeroptera, Plecoptera, and Trichoptera (Sarver et al. 2002 cites - Missouri Department of Natural Resources 2001).
>
> Biotic Index quantifies the invertebrate community as to its overall tolerance to organic pollution by summarizing tolerances of individual taxon. The overall pollution tolerance of the macroinvertebrate community is expressed as a single value between 0 and 10, with higher values indicating increased tolerance.

Shannon Diversity Index is a measure of community composition which takes into account both richness and evenness. Ecologists commonly make the following assumptions concerning diversity: 1) a more diverse community is a healthier community; 2) diversity increases as the number of taxa increase; 3) and the distribution of individuals among those taxa should be evenly distributed.

Integrating the RAM macroinvertebrate dataset with our HTI and plotting the results reveals that as the HTI increases (more threat) each of the four macroinvertebrate indices exhibit a reduction in macroinvertebrate community health. This relationship is most apparent for the Headwater/Creek class of streams (Figure 21). Results reveal that our index (HTI) for Headwaters/Creeks does a good job of predicting both the EPTT Index and the Biotic Index. In each plot regression lines generally indicate that macroinvertebrate community health/condition decreases with increasing threat. The relationship between the HTI and Shannon Diversity Index is least clear.

The relationships between each of the four macroinvertebrate indices and our HTI are less clear on streams classified as Small River and larger. This may be because of differences in the impact that local versus watershed conditions exhibit on the local macroinvertebrate community. Despite the “fuzziness” of these plots regression lines still imply that macroinvertebrate community health decreases with higher HTI.

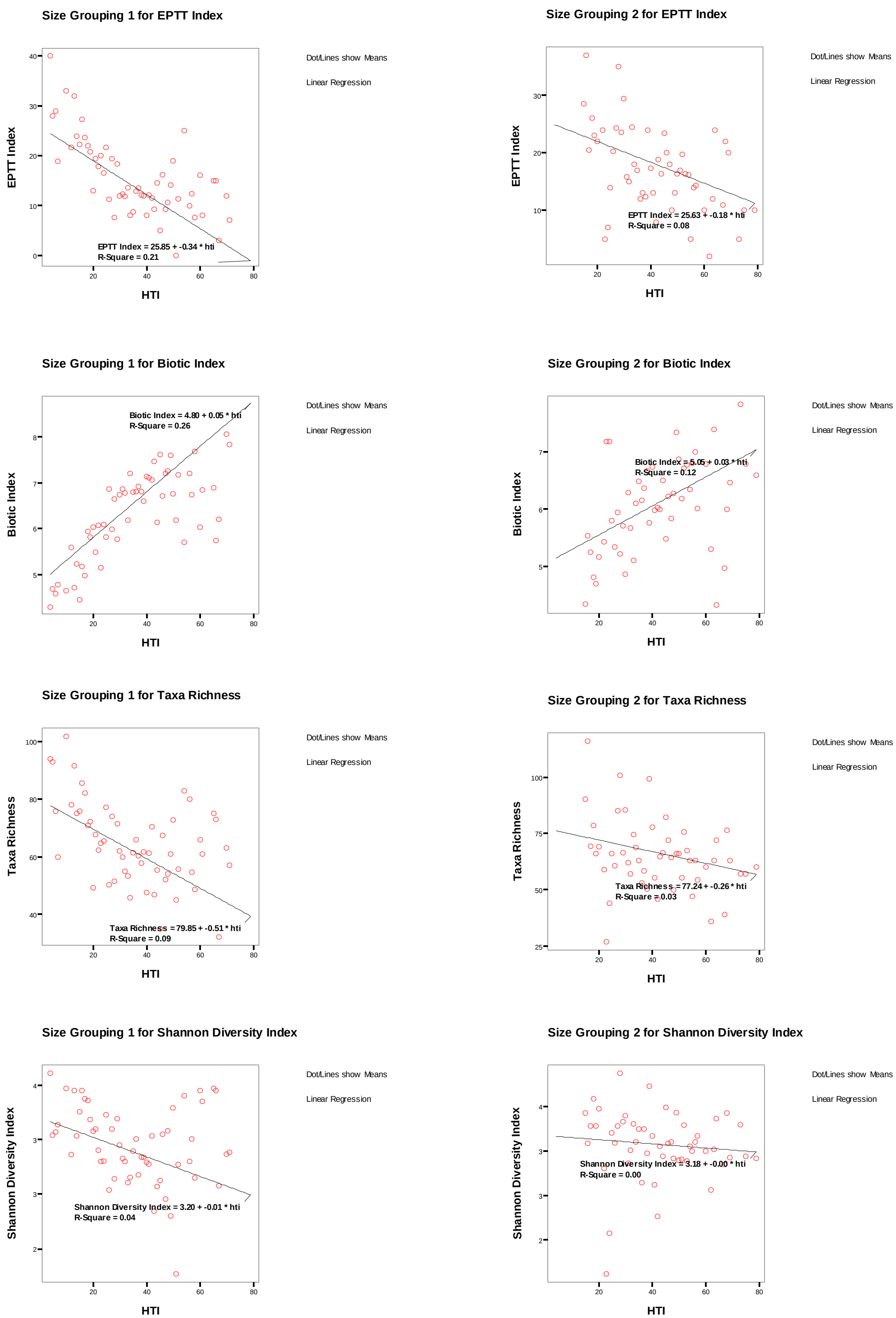


Figure 21. Macroinvertebrate indices plotted against HTI. HTI scaled to stream size groupings (i.e. Size Grouping 1 = Headwaters/Creeks and Size Grouping 2 = Small Rivers and larger.

Classification and Regression Tree Analysis and Modeling (macroinvertebrates)

We also utilized Classification Tree add-on of SPSS version 14.0 to look at the importance of natural character and threats to the macroinvertebrate indices.

Classification and regression tree (CART) analyses are nonlinear/nonparametric modeling techniques that produce either classification or regression trees, depending on whether the dependant variable is categorical or numeric. Classification and regression trees are well suited to analyzing complex ecological data (De'Ath and Fabricius 2000). CART typically employs a recursive-partitioning algorithm which repeatedly partitions the input dataset into a nested series of mutually exclusive groups, each of which is as homogeneous as possible with respect to the response variable (Olden and Jackson 2002). The first split in the tree explains the most variation in the response or dependent variable. The same partitioning algorithm is used to further split the nodes into a series of "child" nodes (De'Ath and Fabricius 2000). The resulting tree-shape output represents sets of decisions or rules for the classification of a particular dataset. These rules can then be applied to a new unclassified dataset to predict which records or, in our case, location will have a given outcome.

We produced CART trees using ecoregion (Bailey's Divisions), natural variables (soils and relief), and threats to see which variables were the most important predictors for each macroinvertebrate index (Figure 22). Examining the trees revealed that the majority of nodes in each tree were comprised of threats, not natural variables. This indicates that for macroinvertebrates threats explain more about the community condition than do natural variables like soils, landform, or ecoregion. Of the threats, crop pesticide applications were the most important predictor for each of the four indices (Taxa Richness, EPTT Index, Biotic Index, Shannon Diversity Index).

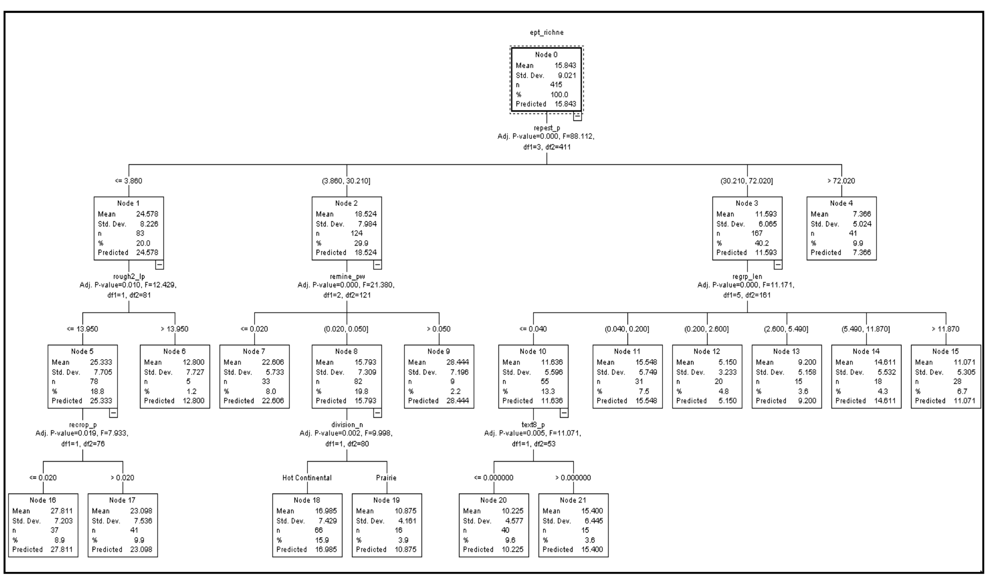


Figure 22. Example regression tree.

To test the individual threats for their utility in predicting or describing ecological integrity we ran a CART model in SPSS Version 14 using only our individual threats to predict EPT Index values and Biotic Index values, the two indices that showed the best relationship to our HTI (Figures 23, 24 and 25). Results indicate that those threats can be used to model/predict the EPT Index and Biotic Index.

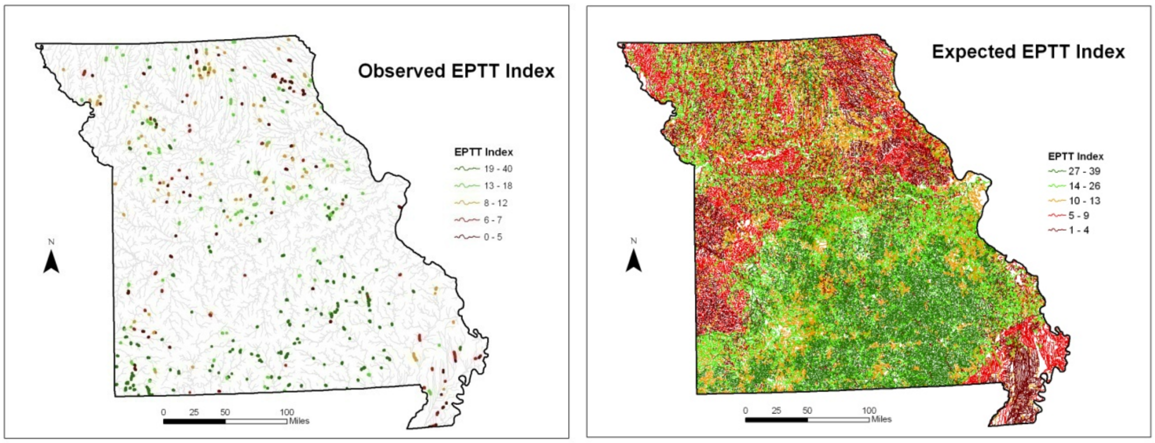


Figure 23. Observed and expected EPTT Index values in Missouri. The expected EPTT Index was modeled using individual threats exclusively in conjunction with the observed EPTT Index values from 415 locations.

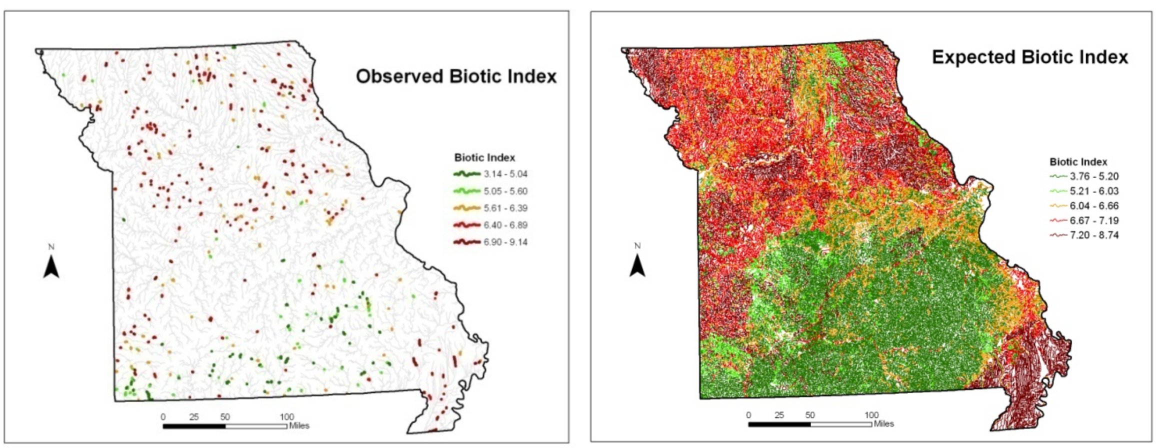


Figure 24. Observed and expected Biotic Index values in Missouri. The expected Biotic Index was modeled using individual threats exclusively in conjunction with the observed Biotic Index values from 415 locations.

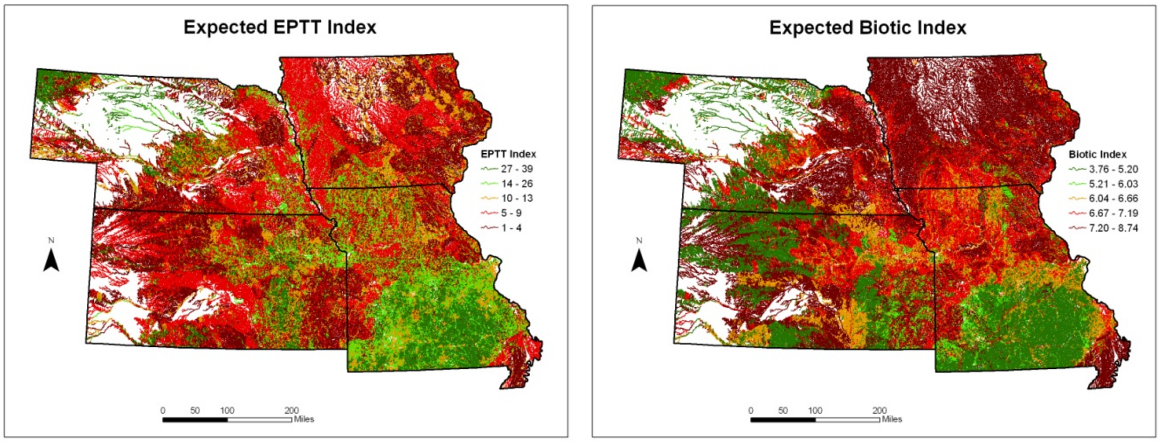


Figure 25. Expected EPTT Index and Biotic Index in EPA Region 7. Modeled using individual threats only. It should be noted that observed EPTT Index and Biotic Index values used to drive the models were only available from sites in Missouri. The resulting model was applied throughout all of EPA Region 7.

Comparing the differences between the observed and predicted/expected indices by subtracting the values and displaying the results via histogram reveals that most predicted values are very close to the observed value; EPT mean of 0.0008 with a standard deviation of 4.19 and Biotic Index mean of -0.0004 with a standard deviation of 0.57 (Figure 26). Acquiring an independent dataset would further strengthen the model validation.

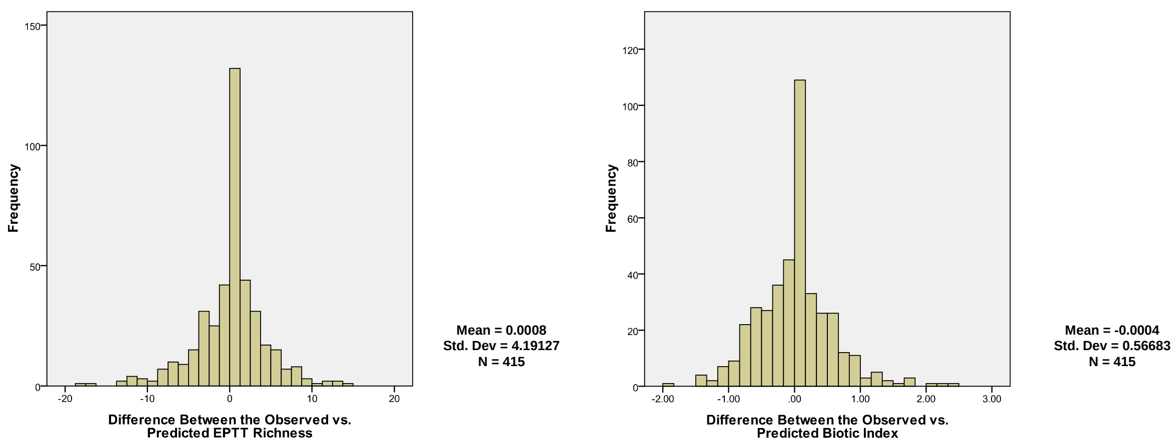


Figure 26. Histograms of the observed minus predicted EPTT Index and observed minus predicted Biotic Index. Values closer to zero indicate a closer correspondence between the observed and predicted indices.

Although these analyses are fairly crude and preliminary they do indicate that our quantified threats and HTI can be used to infer or even predict macroinvertebrate community health for stream segments in Missouri and even EPA region 7. These analyses would certainly benefit from incorporating macroinvertebrate data from Iowa, Kansas, and Nebraska.

HTI Comparison to Fish Indices of Biological Integrity (IBI)

We conducted the same general analyses as we did with macroinvertebrates, but using fish indices of biological integrity (IBI). The structure of the fish community can be used to characterize the health or ecological integrity of a stream (Karr 1981; Rabeni et al. 1997; Karr and Chu 1999), but accurate characterizations depend on the quality of the data which is impacted by gear bias and sample variance (Rabeni et al. 1997). An IBI is a multimetric approach that combines values from individual metrics, which typically vary regionally to best fit local conditions, into an index to measure biological condition.

As stated previously, our original intent was to utilize data collected for us by the Missouri Department of Conservation at nine new sites in Missouri. However, to increase our validation set and with the help of Matt Combes from the Missouri Department of Conservation, we acquired all available Regional Environmental Monitoring and Assessment Program (REMAP) fish collection data from each of the four states in EPA Region 7. The U.S. EPA initiated the REMAP in EPA Region 7 in 1994. The state specific data was acquired from: Iowa (Iowa Department of Natural Resources), Kansas (Central Plains Center for Bioassessment), Nebraska (Nebraska Department of Environmental Quality), and Missouri (Missouri Department of Conservation). These data were all collected using REMAP protocols, although these protocols varied somewhat by year of collection and state. Collections were made using a combination of electrofishing and seining. Dates of collection ranged from 1994 through 2007.

The raw REMAP fish collection metrics were merged into a single dataset covering EPA Region 7 and joined to our stream segments in a GIS which yielded 1,164 locations with a computed IBI (Figure 27). This equated to 162 locations in Iowa with dates ranging from 2002-2006, 99 locations in Kansas ranging from 1994-2001, 563 locations in Missouri ranging from 1994-2007, and 340 locations in Nebraska ranging from 1994-2007. Nine versions of a fish Index of Biological Integrity (IBI) were computed consistently over the entire study area by calibrating the IBI's to "Fish Region" (Uplands, Lowlands, or Plains).

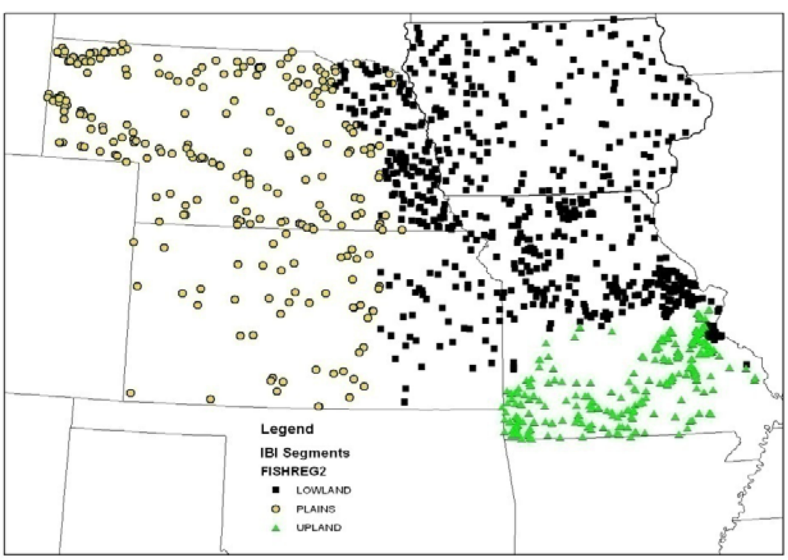


Figure 27. Map showing 1,164 fish IBI locations displayed by Fish Region throughout EPA Region 7.

Although each of the computed IBIs were very similar with regard to values and the relationship with our HTI, one index "IBINEB", generally exhibited a slightly better, albeit weak, relationship to our HTI. Therefore most analyses were conducted using the IBINEB. The IBINEB is an early IBI developed in REMAP for monitoring in Nebraska (Matt Combes personal communication). It is a nine metric IBI created by summing the following and dividing by 0.9. We will refer to the IBINEB as simply "IBI".

1. Native species richness score
2. Native cyprinid species richness score
3. Number of individuals score
4. Sensitive species richness score
5. % tolerant score
6. % omnivore + herbivore score
7. % non-natives score
8. % carnivores score
9. % insectivores + invertivores score

Plotting IBI versus HTI revealed only a very weak linear relationship between these two indices (Figure 28). Although it is not entirely surprising that a weak linear relationship was observed between the HTI and IBI, the lack of a wedge shape pattern in the plot with declining IBI beyond a certain HTI threshold indicates that additional work along the lines of Wang et al. (2008) and Brendon et al. (2008) to establish variable weights may be required.

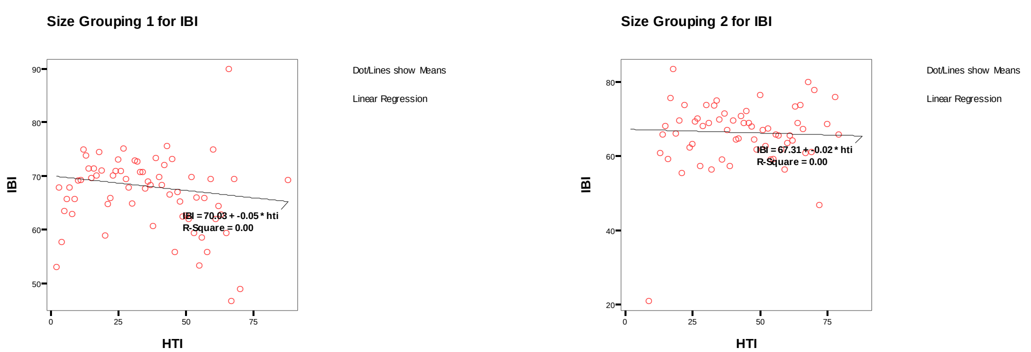


Figure 28. Fish IBI plotted against HTI. HTI scaled to stream size groupings (i.e. Size Grouping 1 = Headwaters/Creeks and Size Grouping 2 = Small Rivers and larger.

Classification and Regression Tree Analysis and Modeling (fish)

Again, we utilized the Classification Tree add-on of SPSS version 14 to look at the importance of natural character and threats to the fish IBI. Examining CART trees revealed that for fish, both ecoregion (Bailey's Division) and physical character (soils and relief) variables are more important determinants of fish IBI than are threats. Other studies have shown similar relationships (Brazner et al. 2007a; Brazner et al. 2007b).

Despite the weak association between our quantified threats and fish IBI we elected to produce a CART model using only our individual quantified threats in conjunction with observed fish IBI values from 1,164 locations (Figure 29). Although done primarily for illustrative purposes, we believe that with additional refinements it may be possible to predict fish IBIs using the datasets developed as part of this project.

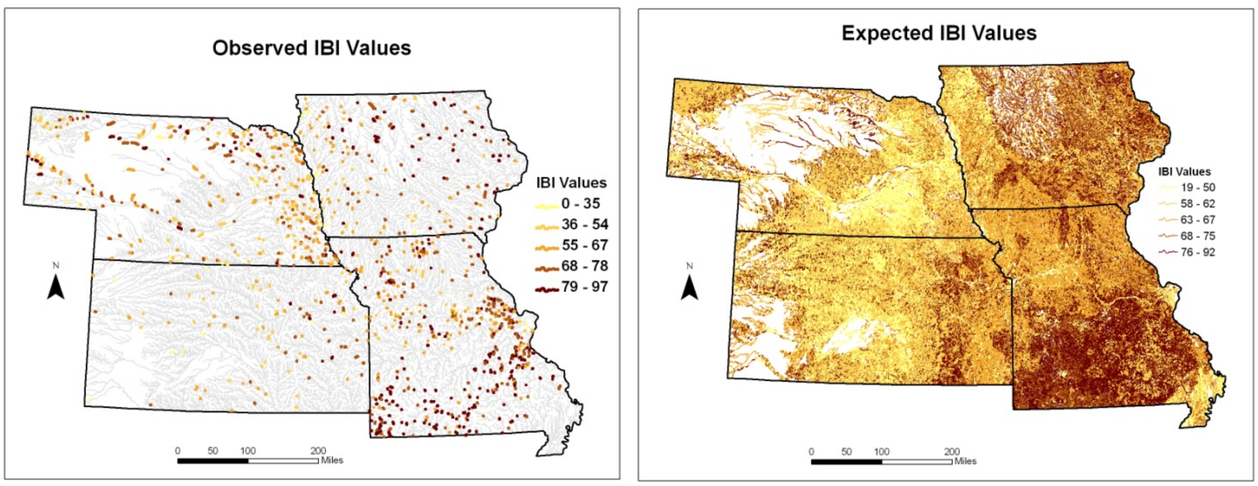


Figure 29. Observed versus expected fish IBI values in EPA Region 7. The expected IBI was modeled using individual threats exclusively in conjunction with the observed IBI values from 1,164 locations. Note that this is very preliminary analysis and would benefit from additional refinement. In fact, our other analyses indicated that threats alone are probably not a good predictor on fish IBI.

To help assess how well the resulting modeled IBIs correspond to known fish IBIs we compared the differences between the observed and predicted fish IBI by subtracting the values and displaying the results via histogram. The computed differences exhibit a mean value of -0.0008 and standard deviation of 9.83 (Figure 30).

These relatively crude analyses indicate that the threats quantified as part of this project are probably not suitable for accurately modeling fish IBIs. Additional work is needed to examine relationships between ecoregion, physical character of the stream/watershed, and threats to fish IBIs.

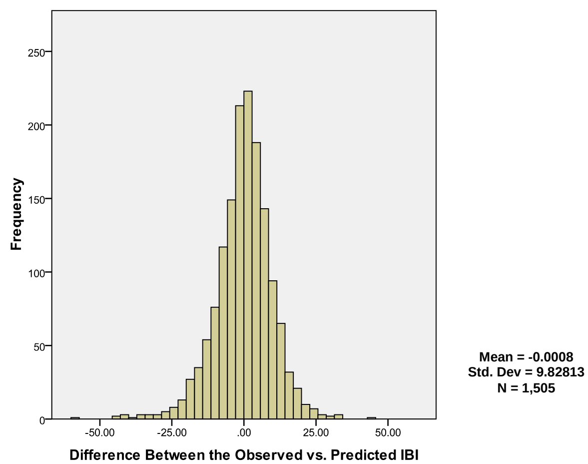


Figure 30. Histogram of the observed minus predicted fish IBI. Values closer to zero indicate a closer correspondence between the observed and predicted IBI.

Human Threat Index (HTI) Discussion

The mean overall HTI score for EPA Region 7 as a whole is 29. Considering the mean HTI for each state individually reveals that Iowa had the highest HTI (more threat) at 38 followed by Missouri (31), Kansas (27), and Nebraska (25). These mean HTIs could easily be computed for any ecological or political unit allowing direct comparisons to be made. The resulting HTI highlights areas with high human use demands. Iowa stands out as having relatively consistently high potential threat statewide due largely to agricultural land uses and related activities. In Missouri, the Ozarks as a region tends to have less overall potential threat, while both Kansas and Nebraska have highest threats in the east where population densities and agricultural activities are most intensive.

Our resulting human threat index (HTI) is by no means a substitute for more detailed site assessments of ecosystem health. There was limited empirical validation of the resulting HTI using biological indicators from fish and macroinvertebrates which in turn have limitations regarding methodologies and the establishment of reference conditions. The HTI was developed as a relative measure of ecological integrity of the stream resources within EPA Region 7 and to provide insight into the spatial distribution of the various anthropogenic threats across the region. More research is needed on how specific threats and stressors influence the ecological integrity of receiving waters.

Although the human threat index (HTI) was not necessarily intended to predict or infer multimetric biological indicators like IBIs, these indices are frequently used to gage stream ecological integrity. We recognize that there are other more robust approaches to index validation, but the approach taken using macroinvertebrates indicates that our overall human threat index does reflect a gradient in biotic integrity over EPA Region 7 and has the potential to be used to predict macroinvertebrate community indices. More work is needed to explore the relationship between our HTI and fish IBIs. Future work may be able to subject our index to more rigorous validation and evaluation, perhaps lending credence to index revision in the future.

As Brazner et al. (2007) acknowledge, species respond to their environment in ways controlled by their body size with larger organisms utilizing broader home ranges thereby experiencing their environment over larger spatial scales. Based on our HTI's correspondence to macroinvertebrates and lack of correspondence to fish IBIs this may indicate that our HTI is more sensitive to local conditions than watershed conditions.

The resulting threat metrics show promise for modeling biological indices for both macroinvertebrates and fish. In our analyses we used individual quantified threats as the sole predictor of biological indices to show the utility of this data. These models could be improved by utilizing physical character (i.e. ecoregions, soils, relief) in conjunction with the threats to drive the models.

The strengths and weaknesses of our approach are related to both the assessment process used and the availability and quality of the threat datasets quantified for the assessment. A strength of our approach to developing an HTI is that it incorporates many threats from multiple sources at both the local and watershed scales and combines these into a single measure of potential threat for every stream segment throughout a large geographic area at a relatively fine resolution. Beyond the index itself, the individual quantified threats are useful on their own for generating maps or to isolate distinct stream segments meeting specific criteria.

There are also several weaknesses to our approach. The first is that our resulting index is very much a factor of the threat data that was available consistently over the four state area. In addition, streams beginning outside of Region 7 do not have their full watersheds considered in the analysis which is reflected in the HTI. It is also possible that one single threat could significantly diminish the ecological integrity at any given location. Because our HTI is based on cumulative total threat, locations with a single pervasive threat will invariably score low giving the false indication that ecological integrity is good. Another weakness is that, other than distance weighting, we were generally not able to assign specific weights to the individual threats incorporated into the overall index. It should be noted that different weightings would likely be needed to best account for different components of ecological integrity or various multimetric biological indicators. An extension of this study could attempt to establish relationships, weights, or thresholds among the various threats to one or more components of ecological integrity.

There are many components of aquatic ecological integrity including flow regime, physical habitat, water quality, energy/nutrient dynamics, and biotic interactions. Additional research is needed to determine how well the HTI reflects these components of aquatic ecological integrity and perhaps develop a separate index for the each component of ecological integrity.

During our regional oversight committee meetings we found that many of the participants were more interested in the individual threat data assembled and quantified as part of this project than they were in the resulting HTI. Because there are so many ways of creating an HTI a number of individuals felt that users of the data would be more interested in combining subsets of the quantified threats and applying weights in ways that would most suit their individual management or research needs.

We believe that the datasets developed as part of this project will provide much needed and valuable information to natural resource professionals. It is our hope that natural resource professionals will find the resulting datasets useful for developing management strategies and conducting research throughout EPA Region 7.

Chapter 4

Resulting Data Products and Data Use

Data Products

The data package consists of raw input threat data layers, the assessment units used for quantifying the threats, related dbf tables containing the quantified threat data, the human threat index (HTI) and some supplemental data that was also quantified and intended to supplement the quantified threats (Figure 31). The supplemental data consists of quantified soils, landcover, and relief information about the drainage area above each stream segment. This supplemental data is useful for identifying streams and watershed with similar physical character and can be used in conjunction with the quantified threats and HTI. This data will be served on the MoRAP web page.

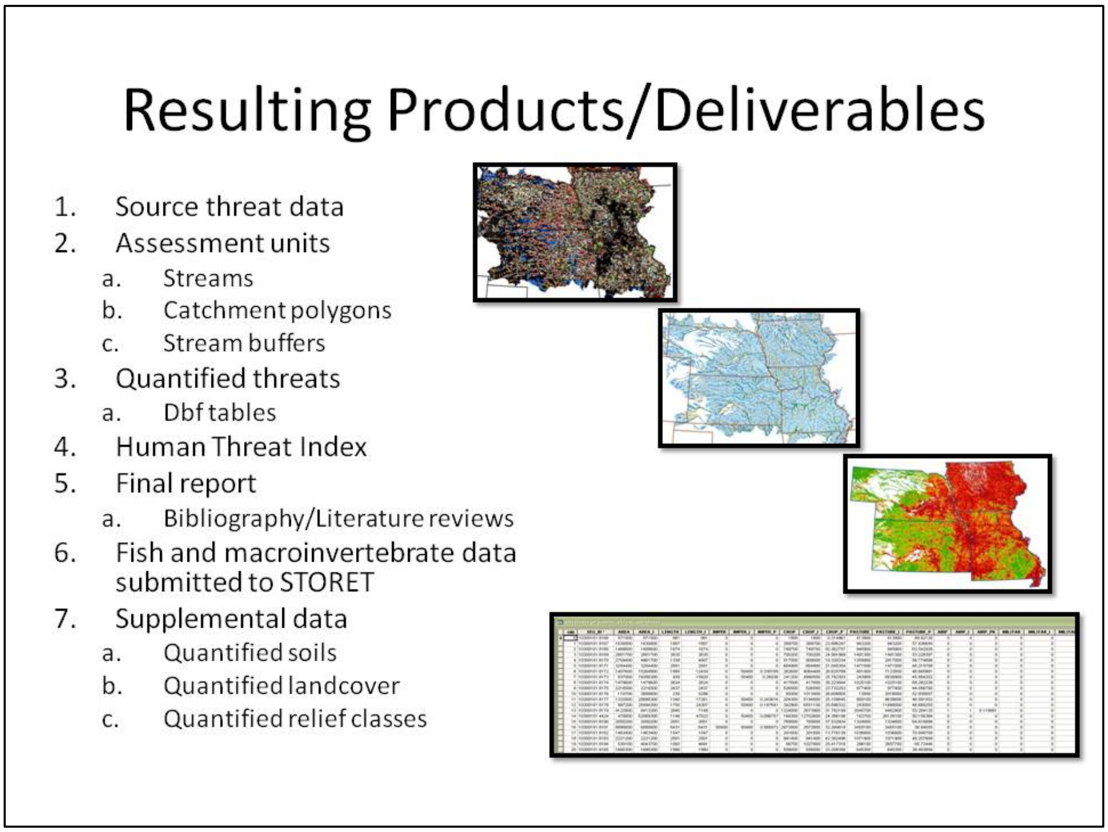


Figure 31. The basic products and deliverables.

The data quantified and developed as part of this project resides in five principal dbf tables.

1. Human_threat_attributes.dbf (the raw quantified threats)
2. Distance_to_threats.dbf (through network distances to select threats)
3. Fragmentation.dbf (fragmentation from major impoundments)
4. Rescaled_weighted_threat_data.dbf (the rescaled and distance weighted threats used to construct the human threat index)
5. Humant_threat_index.dbf (the local, watershed, and overall human threat indices)

Seven additional dbf tables contain information about the physical and vegetative character of each stream segment's drainage area.

1. Landcover.dbf
2. Relief.dbf
3. Riparian_landcover.dbf
4. Soil_hydro_group.dbf
5. Soil_rock_depth.dbf
6. Soil_rock_frag.dbf
7. Soil_texture.dbf

Tables 5, 6, and 7 provide lists of all of the data quantified as part of this project. All of these quantified threats reside in the aforementioned dbf tables that can be related to any of the GIS layer assessment units for query or display.

Table 5. The primary quantified data. All of this data is seamless across state boundaries.

Quantified Data (Seamless State to State)			
1.	Impervious surface	21.	Railroad and stream intersections
2.	Cropland	22.	Waste water treatment facilities
3.	Pasture	23.	Toxic release inventory sites
4.	Airports	24.	Resource conservation recovery information system
5.	Military bases	25.	Estimated kilograms of crop pesticide
6.	Lead mines	26.	Landfills
7.	Coal mines	27.	Headwater impoundments
8.	Dams	28.	Confined animal feeding operations
9.	Road and stream intersections	29.	Dollar amount of livestock sales
10.	Certified water wells	30.	National pollution discharge elimination system sites
11.	Superfund sites	31.	Length of channelized or ditched streams
12.	Major impoundments	32.	Population from the 1990 census
13.	Length of roads	33.	Population from the 2000 census
14.	Oil and gas wells	34.	Population change between 1990 and 2000 census
15.	Mines excluding coal and lead mines	35.	Stream fragmentation
16.	Leaking underground storage tanks		
17.	Pipelines (crude oil)		
18.	Pipelines (refined products/fuels)		
19.	Pipelines (natural gas, propane, etc)		
20.	Length of railroads		

Table 6. Data sets for which distance to threat was computed (minimum, maximum, and mean). All of this data is seamless across state boundaries.

Distance to Threats (Seamless State to State)			
1.	Airports	9.	Superfund sites
2.	Dams	10.	Toxic release inventory sites
3.	Military bases	11.	Waste water treatment facilities
4.	Coal mines	12.	Confined animal feeding operations
5.	Lead mines	13.	Landfills
6.	Other mines	14.	National pollution discharge elimination system sites
7.	Oil and gas wells	15.	Resource conservation recovery information system
8.	Leaking underground storage tanks		

Table 7. Additional supplemental quantified data. This information was provided to help characterize streams and watershed according to physical character. Some components of the riparian landcover (crop and pasture) are considered potential threats.

Supplemental Data	
1.	2001 National Landcover Dataset (16 class)
2.	Riparian Landcover (16 class)
3.	Soil Texture (12 classes)
4.	Soil Hydrological Group (8 classes)
5.	Soil Rock Fragment Volume (6 classes)
6.	Soil Depth to Bedrock (7 classes)
7.	Relief Classes (8 classes)

We made an effort to keep all dbf table field names as simple and intuitive as possible. Generally, there are four fields associated with each quantified threat; an amount for the local catchment polygon, a local amount per unit area, an amount for the entire drainage area above each stream segment, and a watershed amount per unit area. For example, the raw quantified coal mines are represented by the field names "Coal", Coal_lpk, Coal_i", and "Coal_pk". It should be noted that each of these fields has the same prefix 'coal' followed by a different suffix. No suffix indicates that the information in that data field represents the local catchment amount, the suffix '_lpk' is the number of coal mines per square kilometer in the local drainage, the suffix '_i' indicates the amount in the inclusive drainage area for the stream segment, and the suffix '_pk' is the number of coal mines per square kilometer in the drainage above the stream segment. Area features like landcover classes have a similar naming convention. A prefix like "crop" without a suffix indicates the information in that data field represents the amount of cropland in the local catchment, the suffix '_lp' represents the percent of the local drainage area in cropland, the suffix "_i" indicates the amount of cropland in the inclusive drainage area for the stream segment, and the suffix "_p" represents the percent of the drainage area in cropland. Features for which a distance to threat was computed are represented with three fields. Again, a standard prefix (i.e. coal, cafo, lead, etc.) is followed by one of three suffixes. For example the field "Cafo_min" represents the minimum distance to the nearest confined animal feeding operation (CAFO) upstream, "Cafo_max" represents the distance to the furthest CAFO upstream, and "Cafo_ave" represents the average or mean distance to all CAFOs collectively upstream. All distances are measured through the stream channel.

Data Use

The resulting data suite is designed to be very user friendly for individuals with basic GIS skills. All assessment unit layers (streams, catchments, and stream buffers) contain an identifier for each stream segment called "Seg_id". In addition, each dbf table of quantified threats also contains the field "Seg_id". As such, "Seg_id" serves as the common identifier to relate the dbf tables to any of the assessment unit layers and vice versa. All table relations are one-to-one. The key points to using the data are: 1) know what data is available, 2) look at the available metadata, 3) understand the common identifiers, 4) learn by use and start exploring the data suite.

Natural resource professionals will find a number of potential uses for the data developed as part of this project. The data suite is well suited for data inventories and assessments, experimental design, identifying streams with similar threats and similar physical character, permit review and compliance, identifying information needs, and education and outreach.

A tremendous amount of data was quantified for each 1:100,000 stream reach that is "on the shelf" and ready for use in a geographic information system (GIS). The data can be used to generate maps displaying quantified individual threats for the drainage above each stream segment (i.e. distance to nearest upstream coal mine, percent cropland, length of road) (Figure 32 A). By combining multiple tables complex queries of the data can be performed to isolate distinct stream segments with specific criteria (Figure 32 B). The resulting data suite provides a large amount of data that can facilitate gathering statistics for reporting. Finally, the data suite as a whole serves as a 'decision support system' for natural resource management.

When combined with other data sets and professional knowledge this data suite should provide a valuable component to the natural resource professional's "tool kit" to help improve understanding and foster more informed decision making.

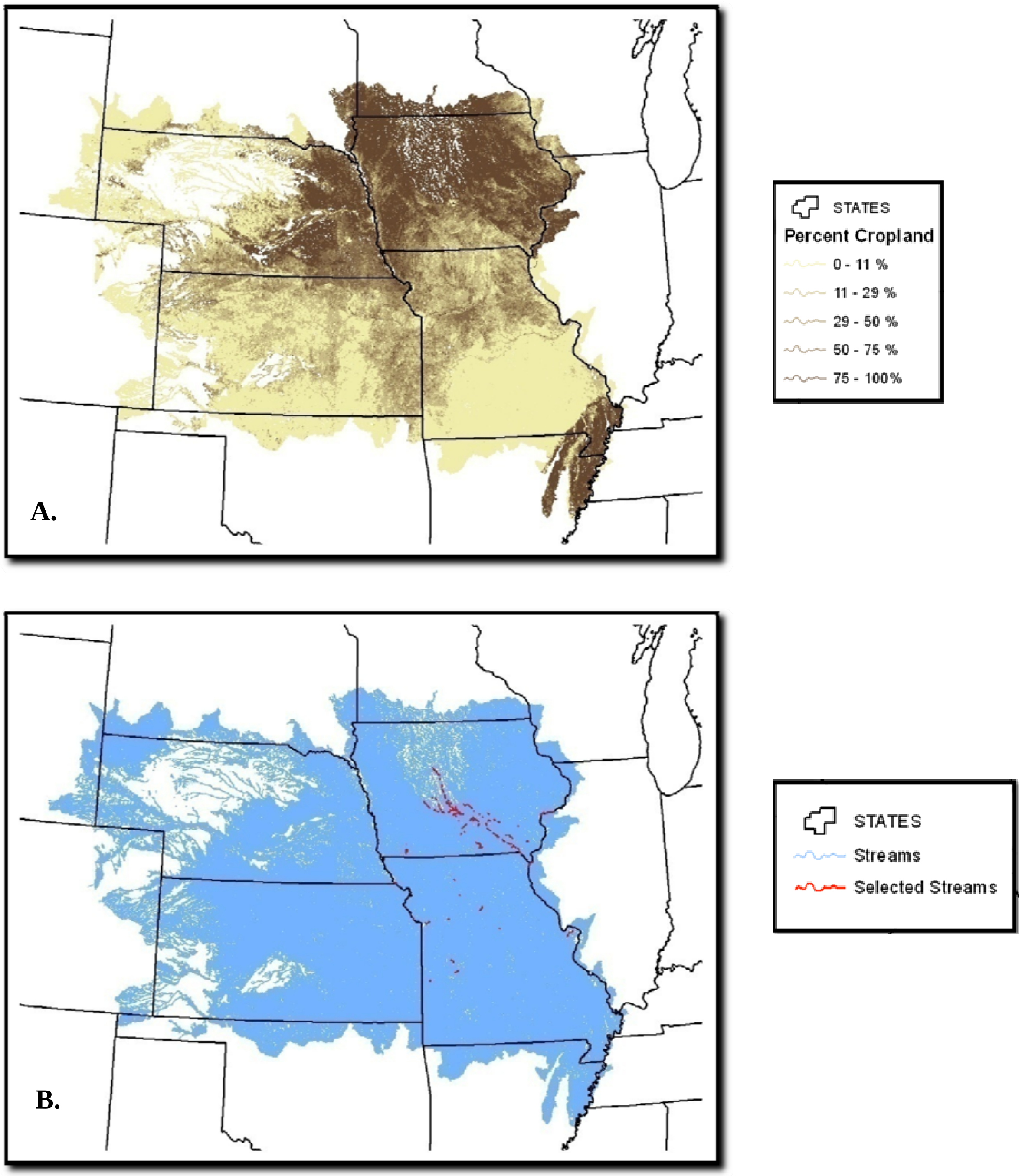


Figure 32. Map A: Percentage of cropland in the drainage area above each stream segment.
Map B: Stream segments with at least 10 coal mines and 10 leaking tanks upstream and with both a coal mine and leaking tank within 10 kilometers upstream.

Data Use Discussion

As Norris et al. (2007) point out, the quality and availability of data limit assessments and even through careful data scrutiny errors are inevitable. As described in Chapter 2 there were many data gaps and limitations related to the availability and quality of the source data used for this project. The three basic issues encountered with source data included: 1) location or horizontal positioning on the landscape, 2) incompleteness, and 3) having multiple sources of the "same" data. Probably the most important data gap or limitation is that this project in no way accounted for every possible threat to aquatic ecosystem integrity. The resulting products from this project are a reflection of data that were identified as being a potential threat and, as importantly, data that were available for use across a large geographic area.

Accounting for human threats within a geographic information system GIS is a difficult task. The metrics used in our human threat index (HTI) rely on only a sample of the potential threats affecting aquatic ecosystems and, as such should be considered only a partial measure of human disturbance. There was relatively limited work done using macroinvertebrates and fish to quantify relations between our HTI, or the individual threat metrics with the ecological integrity of stream resources throughout the EPA Region 7. The HTI was developed as a relative measure of ecological integrity and to provide insights into the spatial distribution of threats over the entire study area.

Suggestions for overcoming some of these data limitations include making use of better source data as these become available in the future. The HTI would benefit from additional refinement and validation using other methods and more and/or different ecological indicators. Acquiring and utilizing macroinvertebrate data covering all of EPA Region 7 would substantially strengthen the results of the threats to macroinvertebrate community condition assessments.

It should also be noted that when creating maps displaying the human threat index (HTI) choices as to the legend and color palate impact the perception people will have of which areas are more or less threatened.

Appropriate Uses:
1. Statewide or region wide planning
2. Large-area resource management planning
3. Research on regional threats to aquatic ecosystem integrity
4. Watershed inventory and assessment
5. Monitoring – selecting reference sites
6. Landowner incentive programs
7. Education and outreach

Inappropriate Uses:
It is far easier to identify appropriate uses than inappropriate ones, but some examples include:
1. Generating precise measurements
2. Precisely quantifying the abundance, health, or condition of any feature (threat, biota, or aquatic system/ecosystem)

3. Assuming the data represents a complete and precisely accurate representation of on-the-ground features or conditions
4. Using the data without acquiring and reviewing the metadata and this report.

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Appendix A

Regional Oversight Committee and Other Participants

Gust Annis	Missouri Resource Assessment Partnership
Scott Sowa	Missouri Resource Assessment Partnership
Aaron Garringer	Missouri Resource Assessment Partnership
Cody Wheeler	Corps of Engineers
Benita Hamilton	Environmental Protection Agency (Region 7)
Cathy Wooster-Brown	Environmental Protection Agency (Region 7)
Eliodora Chamberlain	Environmental Protection Agency (Region 7)
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Tom Isenhart	Iowa State University
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Ryan Waters	Kansas Department of Wildlife and Parks
Deb Baker	Kansas Water Office
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Matt Engel	Missouri Department of Conservation
Mike McKee	Missouri Department of Conservation
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Ken Bazata	Nebraska Department of Environmental Quality
Steve Schainost	Nebraska Game and Parks Commission
Vernon Tabor	U.S. Fish and Wildlife Service
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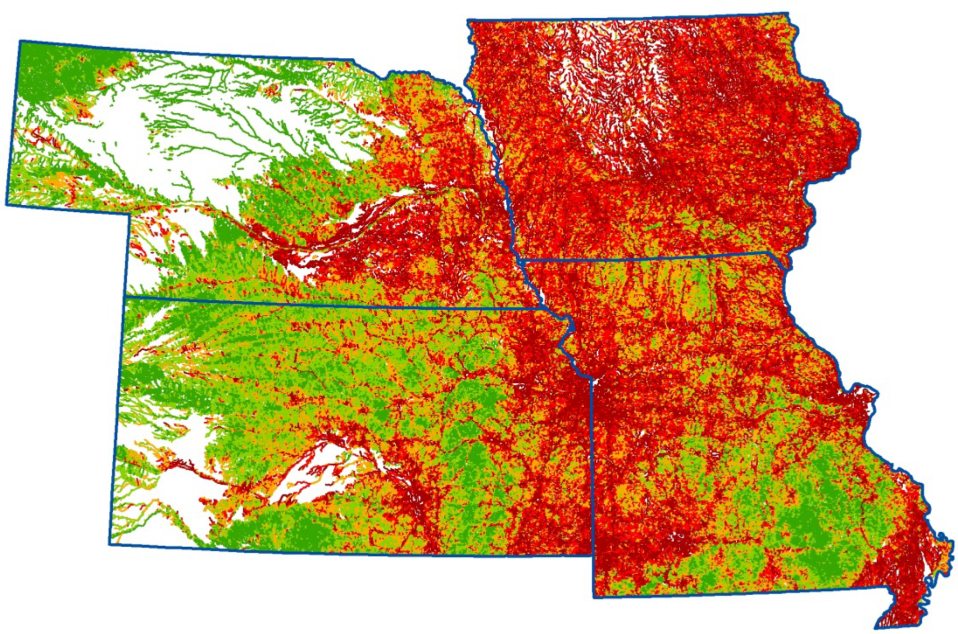
Jo Ellen Hink	U.S. Geological Survey
Kathy Doisy	University of Missouri
Alan Kolok	University of Nebraska, Omaha
Rick Stasiak	University of Nebraska, Omaha
Kyle Hoagland	Water Center, University of Nebraska, Lincoln

Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

Appendix B

Data Layer Descriptions



Streams (R7_Streams)

Description

- This dataset covers most watersheds draining into Region 7; this includes the states of Iowa, Kansas, Missouri and Nebraska. This shapefile consists of consolidated and modified stream shapefiles that were prepared as part of individual Aquatic Gap Analysis Projects for the states of Iowa, Missouri, Nebraska, and Kansas. Individuals from Kansas, Nebraska and Iowa along with staff at the Missouri Resource Assessment Partnership worked cooperatively on these state specific data sets. Generally, this coverage contains selected arcs from the 1:100,000 National Hydrography Dataset (NHD) that was developed by the USGS and EPA. The selected arcs represent the centerlines of wide streams, impoundments, reservoirs, and wetlands as well as the segments of single line streams.

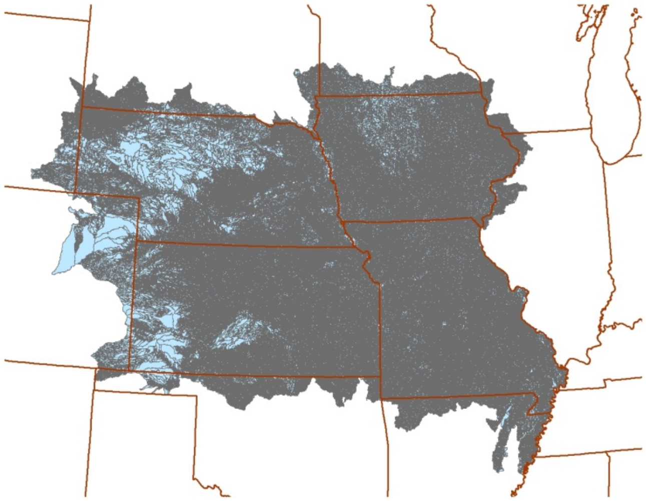
Observations

- Contains all of the information from the EPA Region 7 file.
- This is one of the main datasets used in the analysis for this project.

Source(s)

1. EPA Region 7 Modified Stream Network
   - Who created the data: Missouri Resource Assessment Partnership (MoRAP)
   - Publication date and time: 2006
   - Publisher and place: MoRAP, Columbia Missouri
   - Acquired from: MoRAP
   - Acquisition Date: 2006

Stream Catchments (R7_catchments)



Description

- This dataset is a shapefile representing the catchments for every stream segment in our study area. This dataset was created using the National Hydrography Dataset Plus (NHD Plus) DEM data.

Observations

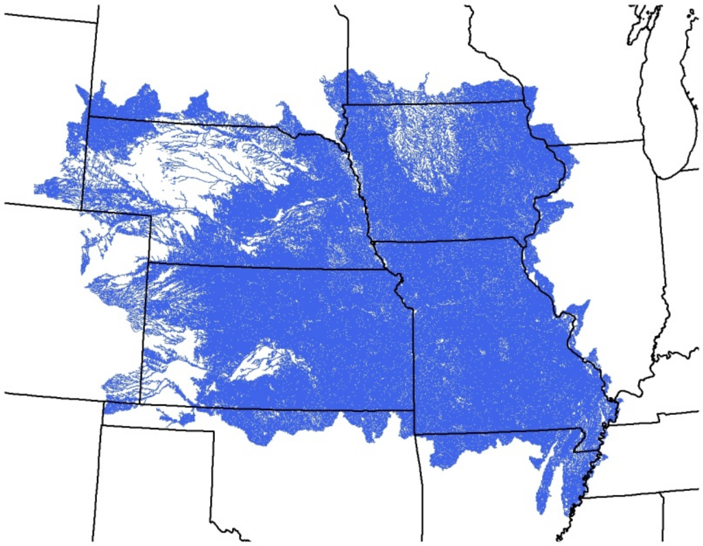
- None.

Source(s)

1. EPA Region 7 Modified Stream Network
   - Who created the data: Missouri Resource Assessment Partnership (MoRAP)
   - Publication date and time: 2006
   - Publisher and place: MoRAP, Columbia Missouri
   - Acquired from: MoRAP
   - Acquisition Date: 2006

2. EPA Region 7 DEM's
   - Who created the data: Environmental Protection Agency & United States Geological Survey, NHD Plus
   - Publication date and time: 2006
   - Publisher and place: Unknown
   - Acquired from: NHD Plus Website
   - Acquisition Date: 2006

Stream Riparian Buffers



Description

- This dataset is a layer representing stream buffers for every primary channel stream segment in our study area (most streams draining into EPA Region 7). This dataset was created using a subset of the R7_streams shapefile. This data layer was created to quantify landcover within riparian areas.

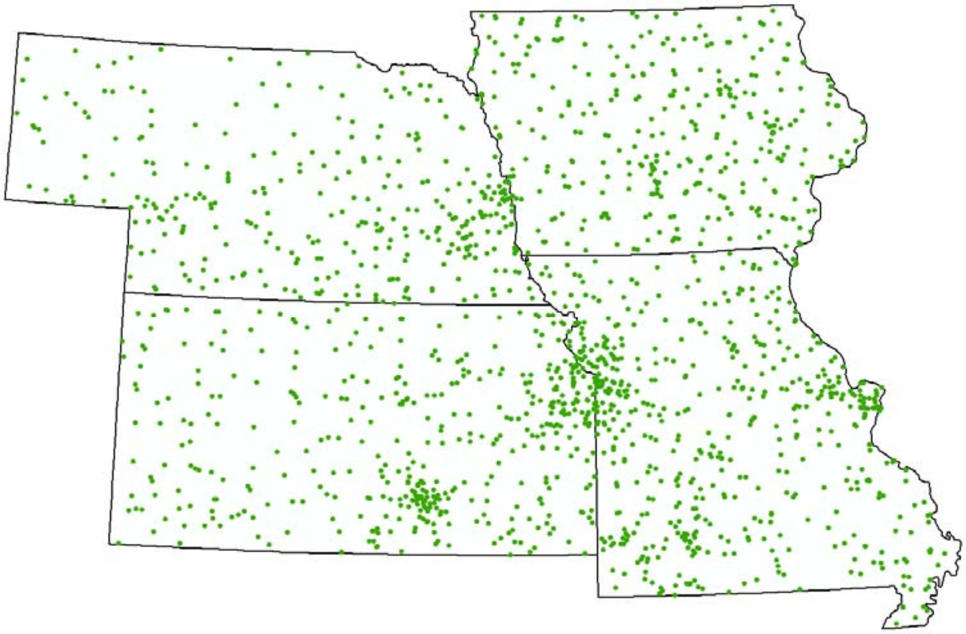
Observations

- Headwaters and Creeks were buffered by 45 meters on a side.
- Small and Large Rivers were buffered by 105 meters on a side.
- Great Rivers (Missouri and Mississippi Rivers) were buffered by 105 meters from the stream bank.

Source(s)

1. EPA Region 7 Modified Stream Network
   - Who created the data: Missouri Resource Assessment Partnership (MoRAP)
   - Publication date and time: 2006
   - Publisher and place: MoRAP, Columbia Missouri
   - Acquired from: MoRAP
   - Acquisition Date: 2006

Airports

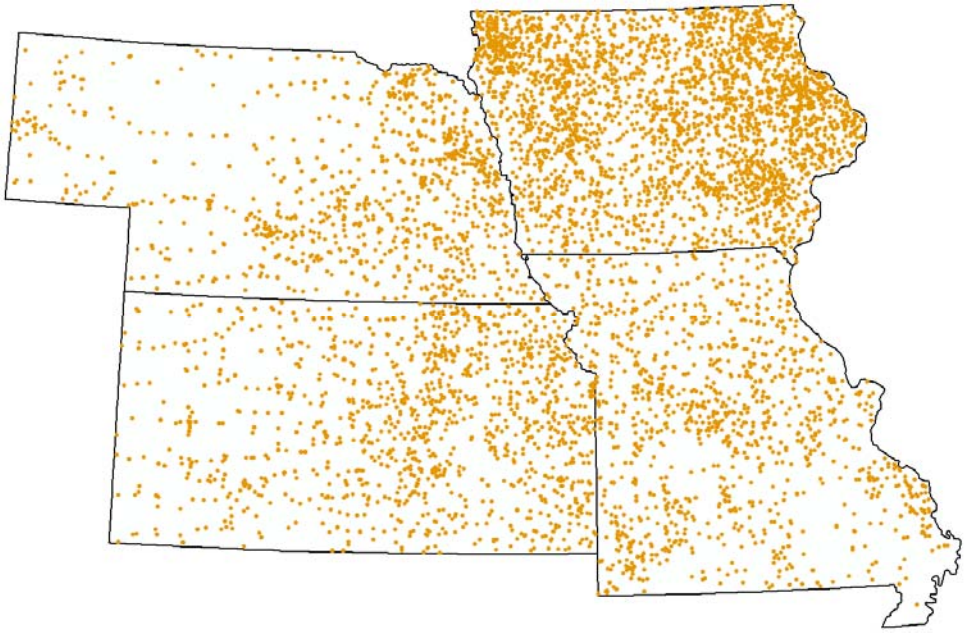


Description
- Airport locations from GDT Dynamap/2000.

Observations
- Data was provided by the EPA and is consistent for all of EPA Region 7.
- This dataset includes private and public airports of all sizes.

Source(s)
1. EPA Region 7 Airports
   - Who created the data: Dynamap
   - Publication date and time: 2004
   - Publisher and place: U.S. Environmental Protection Agency, Region 7, Kansas City, KS
   - Acquired from: EPA Region 7
   - Acquisition Date: 2007

Confined Animal Feeding Operations (CAFO)



Description

- This data was selected from Dunn & Bradstreet data (2003) by SIC code to try to capture animal feedlots in the EPA Region 7 area.

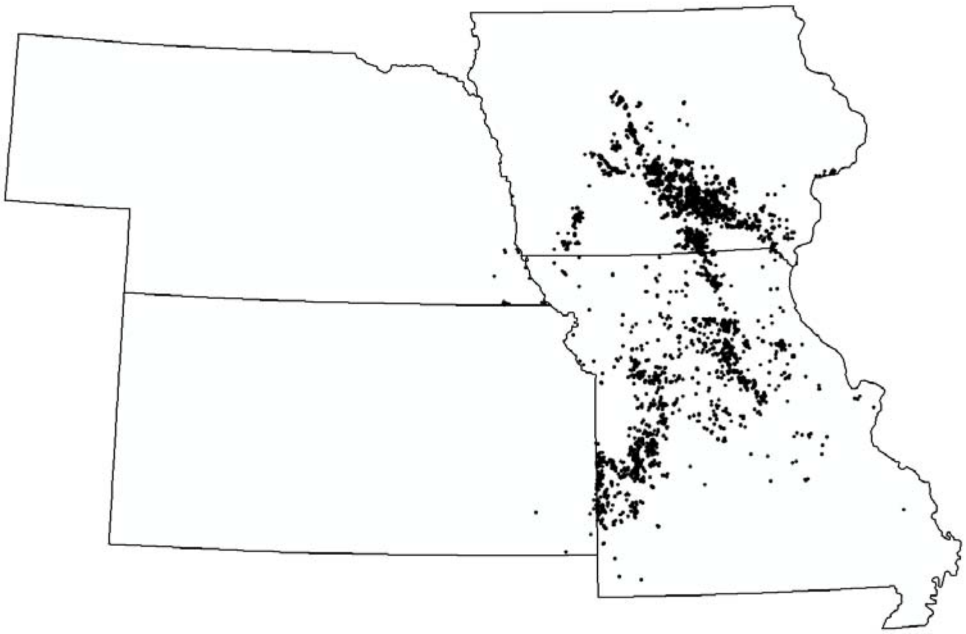
Observations

- One limitation to this dataset is that points representing CAFOs were recorded using one of many methods including: geo-coding facility address, geo-coding owner address, GPS, zip code centroids, etc...

Source(s)

1. EPA Region 7 Confined Animal Feeding Operations
   - Who created the data: Dunn 7 Bradstreet
   - Publication date and time: 2003
   - Publisher and place: U.S. Environmental Protection Agency, Region 7, Kansas City, KS
   - Acquired from: EPA Region 7
   - Acquisition Date: 2006

Coal Mines



Description

- This dataset consists of all coal mines that could be identified using three input datasets; namely the mines datasets provided by the EPA's Better Assessment Science Integrating point & Non-point Sources (BASINS) 2001 data, the Conservation & Survey Division (CSD) University of Nebraska and the Coal Mines of Iowa from the Iowa Department of Natural Resources. We used existing attribution from the aforementioned files to identify coal mines.

Observations

- This dataset is comprised of coal mines that were obtained with the best available data at the time.
- The dataset contains active and abandoned coal mines.

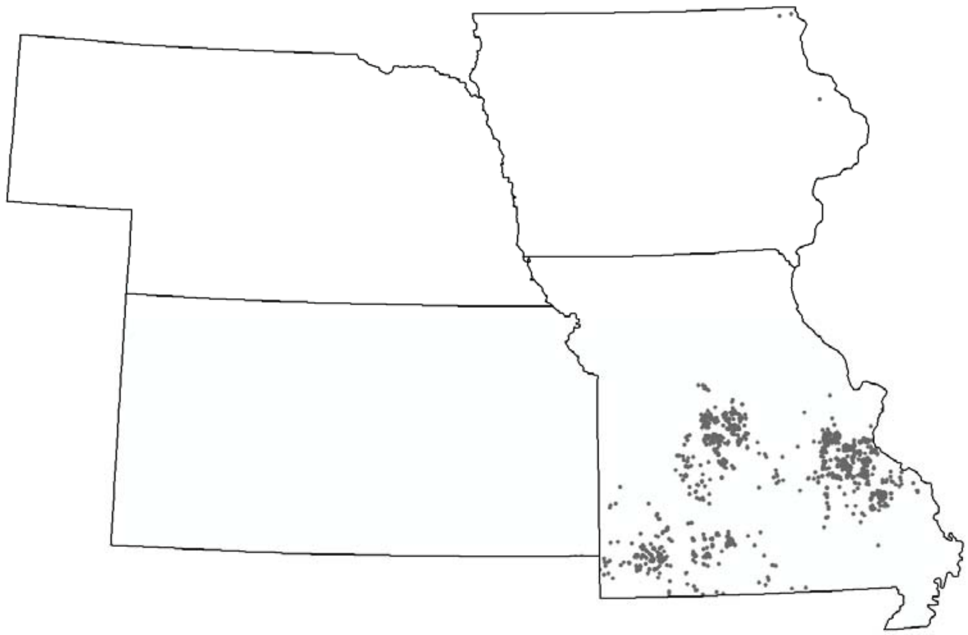
Source(s)

1. EPA Region 7 BASINS Coal Mines
   - Who created the data: Environmental Protection Agency
   - Publication date and time: 2001
   - Publisher and place: U.S. Environmental Protection Agency
   - Acquired from: BASINS Version 3.0 Region 7 CD’s
   - Acquisition Date: Unknown

2. Coal Mines of Iowa
    - Who created the data: Mary R. Howes, Iowa Department of Natural Resources
    - Publication date and time: 8/27/2003
    - Publisher and place: Iowa Department of Natural Resources
    - Acquired from: Iowa GIS Library
    - Acquisition Date: 2006

3. Nebraska Coal Mines
    - Who created the data: Conservation & Survey Division, University of Nebraska - Lincoln (CSD)
    - Publication date and time: 1996
    - Publisher and place: Conservation and Survey Division, University of Nebraska - Lincoln
    - Acquired from: University of Nebraska – Lincoln (CSD)
    - Acquisition Date: Unknown

Lead Mines



Description

- This dataset is a subset from EPA's mines dataset obtained from the EPA's Better Assessment Science Integrating point & Non-point Sources (BASINS) 2001 data. All features that were classified as lead mines were extracted to create this shapefile.

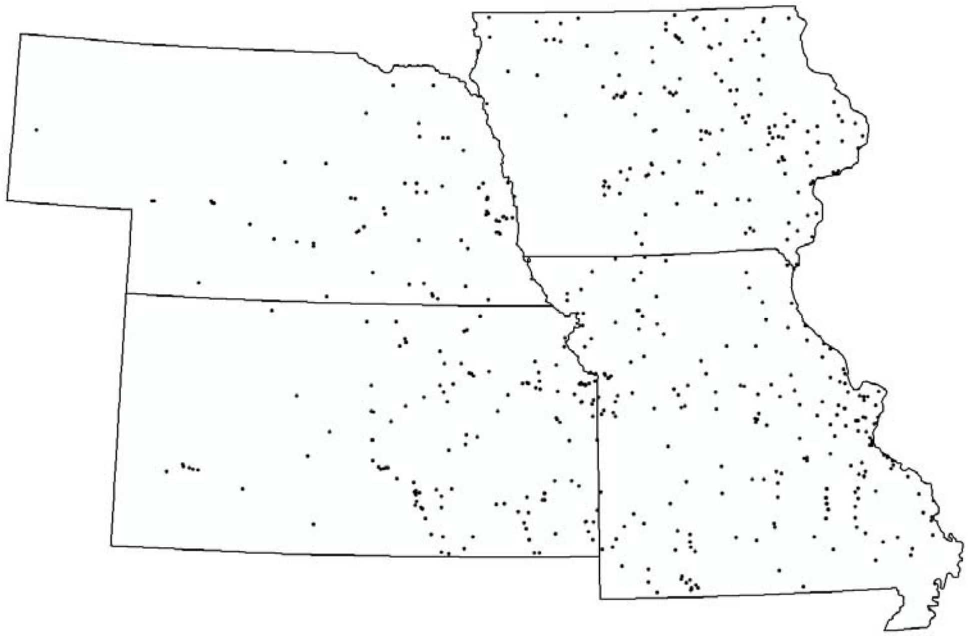
Observations

- This dataset was extracted from the mines dataset from the EPA's Better Assessment Science Integrating point & Non-point Sources (BASINS) 2001 data.
- The data contains both active and abandoned mines.

Source(s)

1. EPA Region 7 BASINS Mines
   - Who created the data: Environmental Protection Agency
   - Publication date and time: 2001
   - Publisher and place: U.S. Environmental Protection Agency
   - Acquired from: BASINS Version 3.0 Region 7 CD’s
   - Acquisition Date: Unknown

Mines



Description

- This dataset is a subset from the USGS mines dataset created by the Minerals Information Team. It is important to note that we removed both coal and lead mines from the original source data. Coal and lead mines were quantified as separate layers.

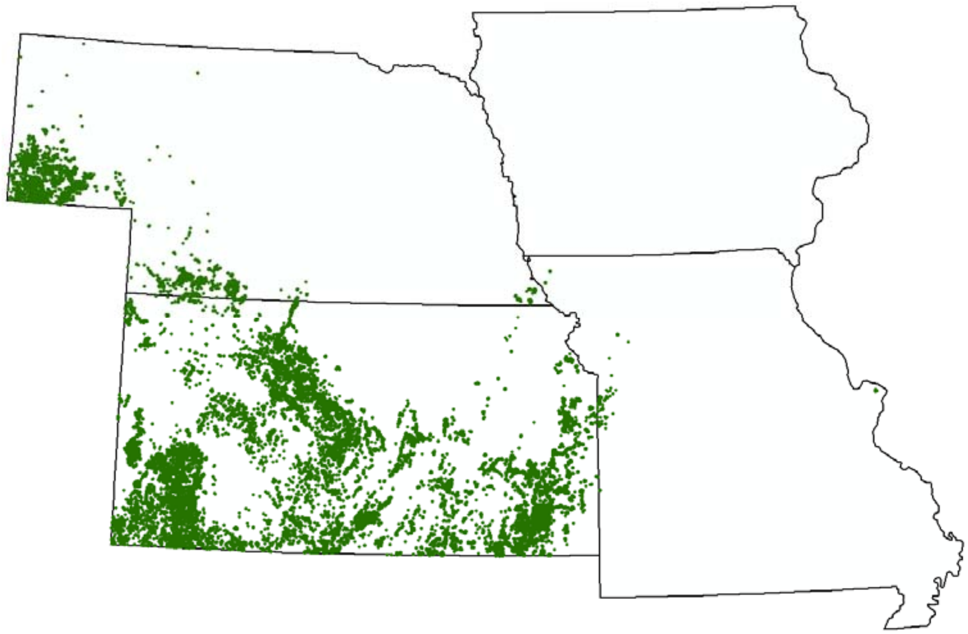
Observations

- The completeness of this dataset is unknown.

Source(s)

1. USGS Mines
   - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS)
   - Publication date and time: 2005
   - Publisher and place: USGS Reston, Virginia
   - Acquired from: USGS Minerals Information Team
   - Acquisition Date: 2007

Oil and Gas Wells



Description

- This dataset consists of all oil and gas wells that could be identified using three input datasets; namely oil and gas well datasets provided by the state agencies of Kansas, Nebraska and Missouri; the state of Iowa did not contain any active oil or gas wells. We used existing attribution from the aforementioned files to identify oil and gas wells that were active and appended the files into one layer.

Observations

- Presently Iowa does not have any active producing oil or gas wells.
- The Missouri file consisted of active oil and gas wells only.
- The Kansas and Nebraska contained all wells that were drilled, plugged, abandoned or active. We restricted this to active wells only.

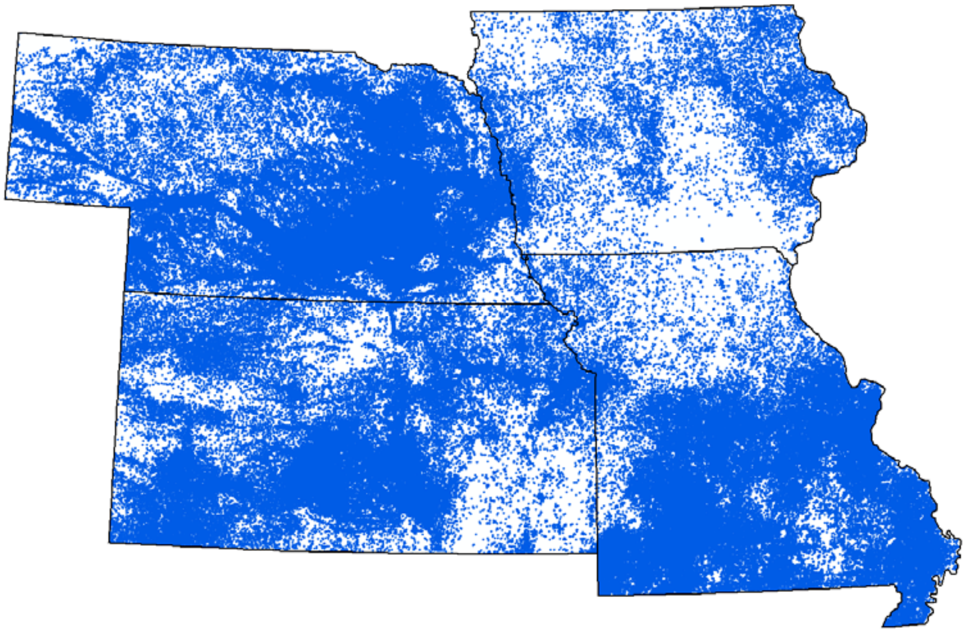
Source(s)

1. Nebraska Oil and Gas Wells
   - Who created the data: Conservation & Survey Division, University of Nebraska - Lincoln (CSD)
   - Publication date and time: 1996
   - Publisher and place: Conservation and Survey Division, University of Nebraska – Lincoln, Lincoln, Nebraska
   - Acquired from: Conservation and Survey Division, University of Nebraska – Lincoln
   - Acquisition Date: 2006

2. Kansas Oil and Gas Wells
    - Who created the data: Kansas Geological Survey
    - Publication date and time: Varies
    - Publisher and place: Unknown
    - Acquired from: Kansas Geological Survey
    - Acquisition Date: 2006

3. Missouri Oil and Gas Wells (Provisional Data)
    - Who created the data: Scott Kaden - Missouri Department of Natural Resources
    - Publication date and time: Unknown
    - Publisher and place: MoDNR, Provisional Data
    - Acquired from: Scott Kaden - MoDNR (Data CD)
    - Acquisition Date: 2007

Certified Wells



Description

- This data set provides information about water wells that are certified by each state in EPA Region 7. Most of the information in these datasets was provided by the well drillers within each state.

Observations

- The well data is made up of four different state datasets.
- Most of the data was created based on township and range legal descriptions.

Source(s)

1. Iowa Certified Wells
   - Who created the data: Mary R. Howes; Iowa DNR
   - Publication date and time: 12/21/1995
   - Publisher and place: Iowa Department of Natural Resources, None
   - Acquired from: Iowa GIS Library
   - Acquisition Date: 2006

2. Kansas Certified Wells
   - Who created the data: Kansas Geological Survey
   - Publication date and time: Varies
   - Publisher and place: Unknown
   - Acquired from: Kansas Geological Survey
   - Acquisition Date: 2006

3. Nebraska Certified Wells
    - Who created the data: Nebraska Department of Natural Resources
    - Publication date and time: 1957-Present at time 060000
    - Publisher and place: Nebraska Department of Natural Resources, Lincoln, Nebraska, United States
    - Acquired from: Nebraska Department of Natural Resources
    - Acquisition Date: 2006

4. Missouri Certified Wells
    - Who created the data: Missouri Department of Natural Resources (MoDNR), Division of Environmental Quality (DEQ), Wellhead Protection Section (WPS)
    - Publication date and time: 1/1/2006
    - Publisher and place: Bob Archer, Geologist, MoDNR
    - Acquired from: Missouri Spatial Data Information Service (MSDIS)
    - Acquisition Date: 2006

Pipelines

(Picture not Available)

Description

- The data for pipelines is sensitive data. EPA Region 7 quantified this data for MoRAP.

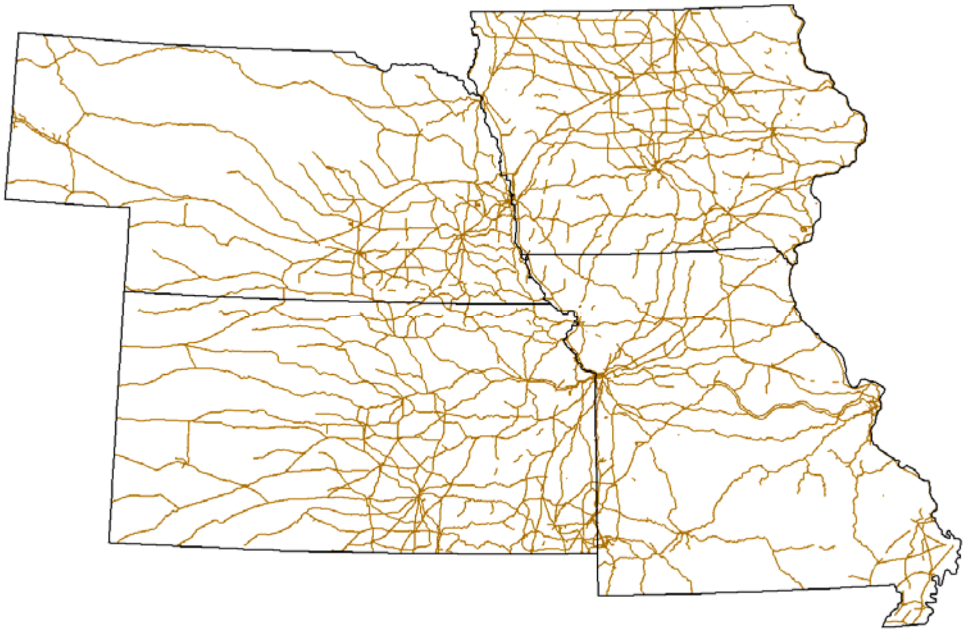
Observations

- The types of files have been split into three different datasets to run the analysis on.
    - The first is “pipeline1” and has the types CRD and crude oil.
    - The second is “pipeline2” and has types diesel and gas, jet fuel, petroleum products, PRD, product, and refined products systems.
    - The third dataset is “pipeline3” and has types HVL products, natural gas, NGL, propane, and propane and ethanol.

Source(s)

1. EPA Region 7 Pipelines (Sensitive Data)
    - Who created the data: Unknown
    - Publication date and time: Unknown
    - Publisher and place: Unknown
    - Acquired from: EPA Region 7
    - Acquisition Date: Never Acquired

Railroads



Description

- Railroads layer from the 1999 TIGER line file dataset.

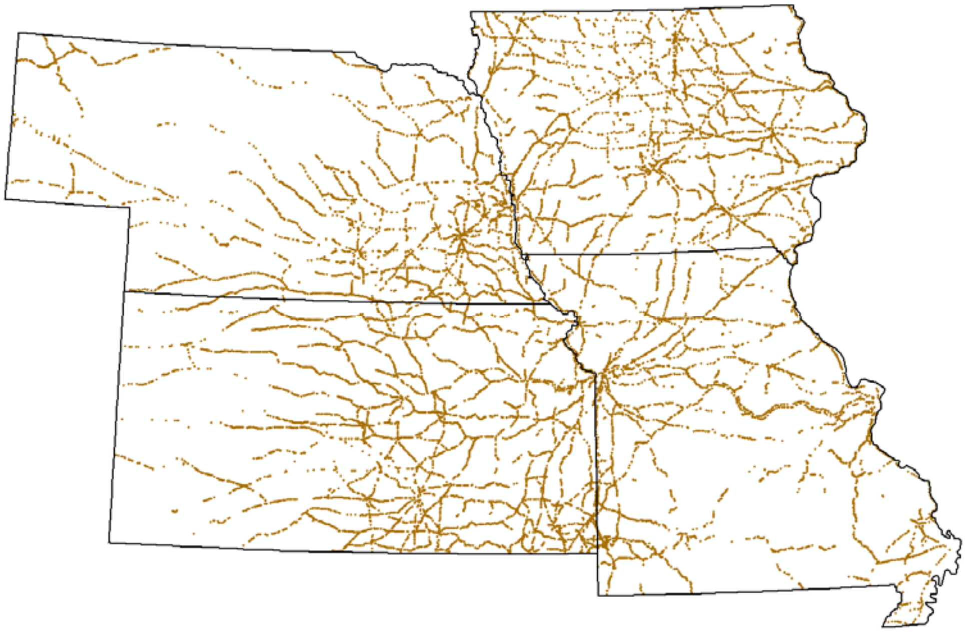
Observations

- None

Source(s)

1. Tiger Line Railroads
   - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division, TIGER/Lines
   - Publication date and time: 2000
   - Publisher and place: U.S. Department of Commerce U.S. TIGER/Line Geography Division, Washington, DC
   - Acquired from: ESRI Website
   - Acquisition Date: 2006

Railroad/Stream Crossings



Description

- This dataset consists of points where railroads cross streams. These locations were identified by intersecting the TIGER/line railroad file with a modified version of the 1:100,000 National Hydrography Dataset (NHD).

Observations

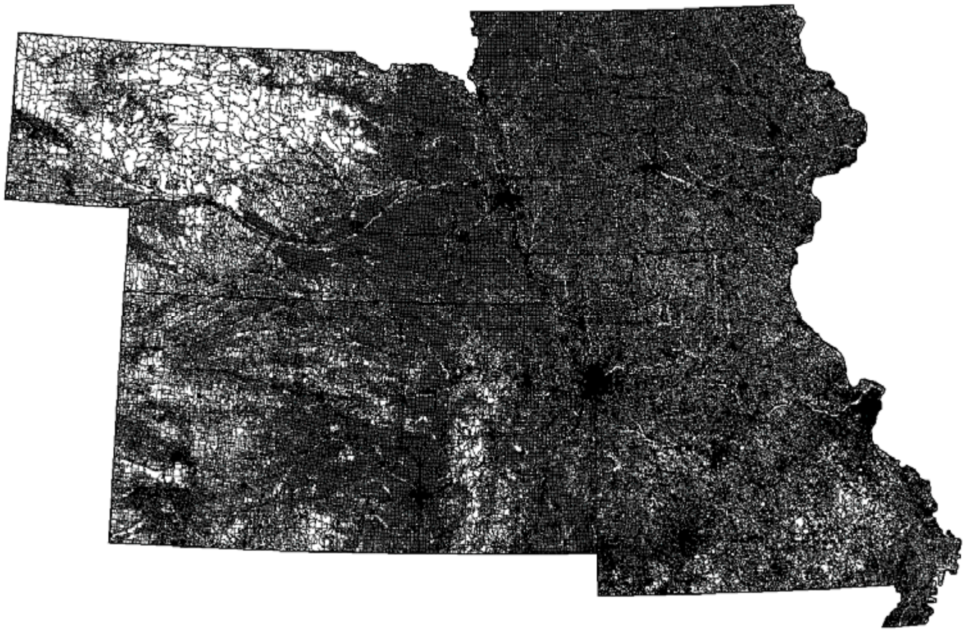
- None

Source(s)

1. Tiger Line Railroads
    - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division, TIGER/Lines
    - Publication date and time: 2000
    - Publisher and place: U.S. Department of Commerce U.S. TIGER/Line Geography Division, Washington, DC
    - Acquired from: ESRI Website
    - Acquisition Date: 2006

2. EPA Region 7 Modified Stream Network
    - Who created the data: Missouri Resource Assessment Partnership (MoRAP)
    - Publication date and time: 2006
    - Publisher and place: MoRAP, Columbia Missouri
    - Acquired from: MoRAP
    - Acquisition Date: 2006

Roads



Description

- Roads layer from the 1999 TIGER line file dataset.

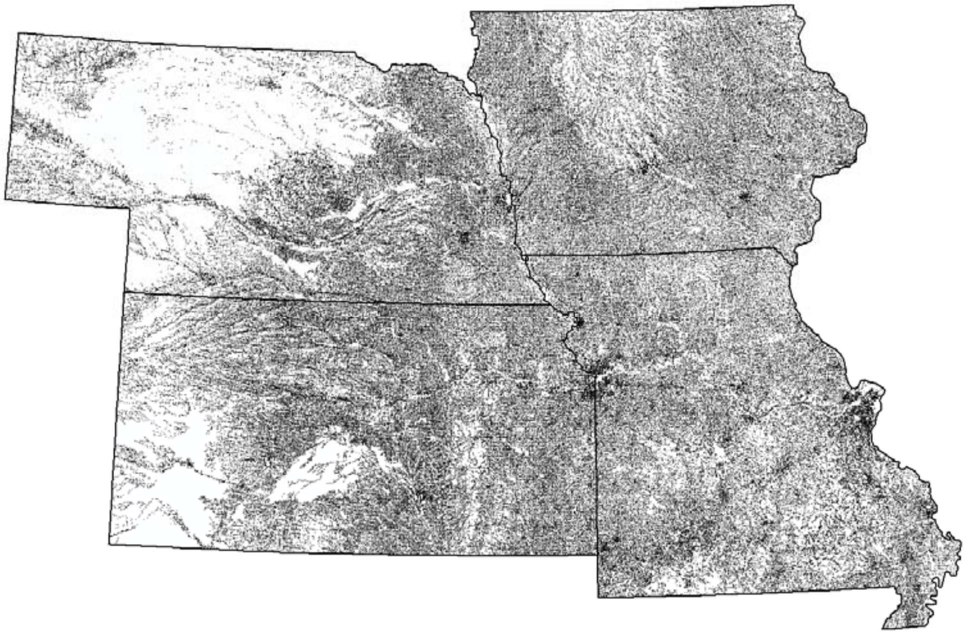
Observations

- None

Source(s)

1. Tiger Line Roads
   - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division, TIGER/Lines
   - Publication date and time: 2000
   - Publisher and place: U.S. Department of Commerce U.S. TIGER/Line Geography Division, Washington, DC
   - Acquired from: ESRI Website
   - Acquisition Date: 2006

Road/Stream Crossings



Description

- This dataset consists of points where roads cross streams. These locations were identified by intersecting the TIGER/line road file with a modified version of the 1:100,000 National Hydrography Dataset (NHD).

Observations

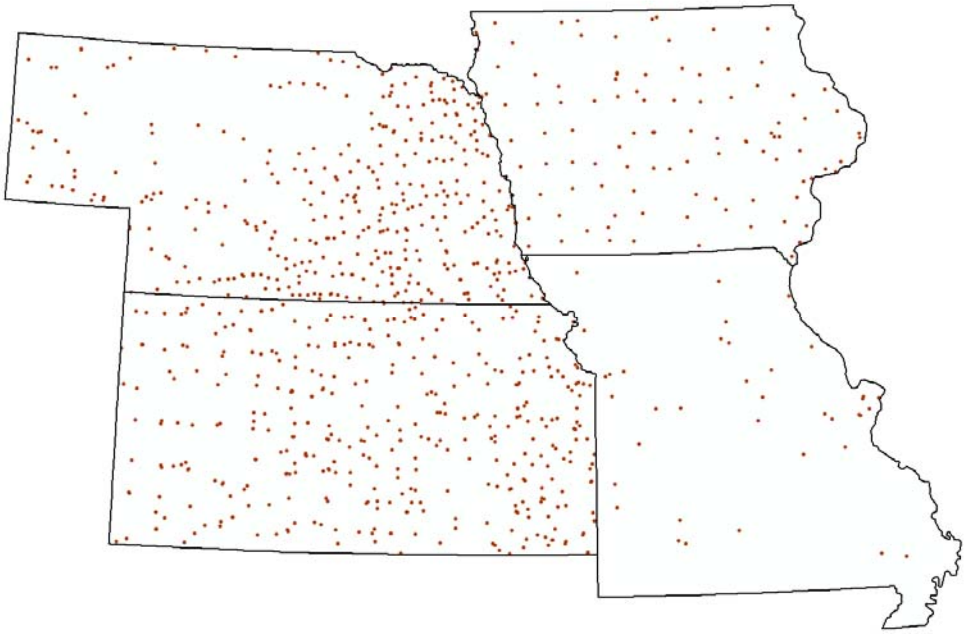
- None

Source(s)

1. Tiger Line Roads
   - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division, TIGER/Lines
   - Publication date and time: 2000
   - Publisher and place: U.S. Department of Commerce U.S. TIGER/Line Geography Division, Washington, DC
   - Acquired from: ESRI Website
   - Acquisition Date: 2006

2. EPA Region 7 Modified Stream Network
   - Who created the data: Missouri Resource Assessment Partnership (MoRAP)
   - Publication date and time: 2006
   - Publisher and place: MoRAP, Columbia Missouri
   - Acquired from: MoRAP
   - Acquisition Date: 2006

Landfills



Description

- We used EPA's BASINS 2001 landfill data for Iowa, Kansas, and Nebraska. The BASINS dataset did not include data for Missouri. A Missouri specific landfill dataset was converted from polygons to points and appended to the EPA BASINS dataset for analysis.

Observations

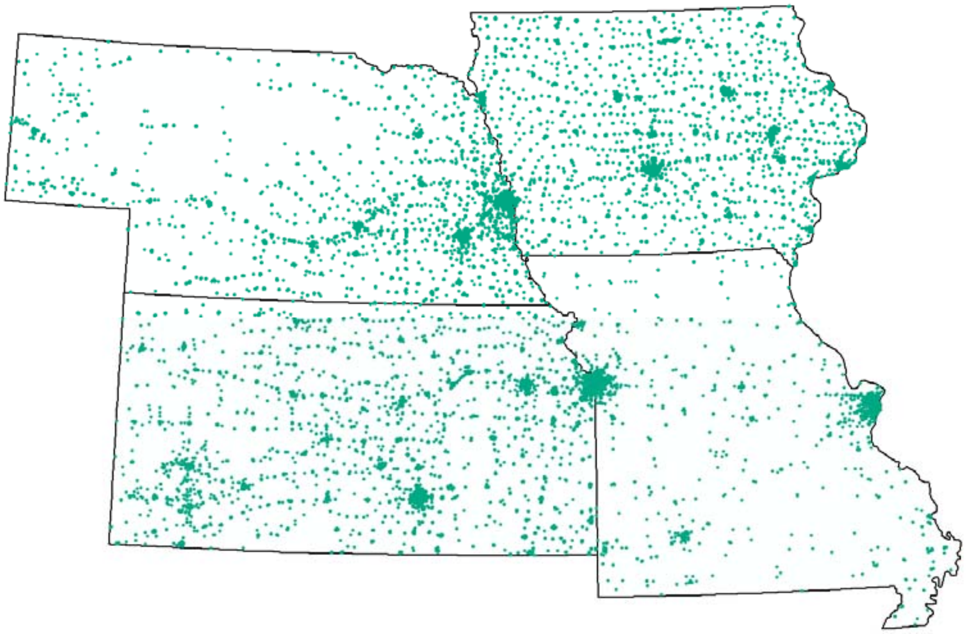
- **None**

Source(s)

1. EPA Region 7 BASINS Landfills
   - Who created the data: Environmental Protection Agency
   - Publication date and time: 2001
   - Publisher and place: U.S. Environmental Protection Agency
   - Acquired from: BASINS Version 3.0 Region 7 CD's
   - Acquisition Date: Unknown

2. Missouri Landfills
   - Who created the data: Missouri Department of Natural Resources, Air and Land Protection Division, Solid Waste Management Program
   - Publication date and time: 2004
   - Publisher and place: MoDNR
   - Acquired from: Missouri Spatial Data information Service
   - Acquisition Date: 2006

Leaking Underground Storage Tanks (LUST)



Description

- This dataset consists of all leaking underground storage tanks (LUST) that could be identified using four input datasets; namely leaking underground storage tank datasets provided by the state agencies of Kansas, Iowa and Missouri; the state of Nebraska did not have a shapefile of LUST sites, but they did have a list with addresses that we were able to geocode. We used existing attribution from the aforementioned files to identify active LUST sites.

Observations

- Various methods for creating the source data include: GPS, zip code centroids, address matching, topographic maps and aerial imagery digitizing, and others.

Source(s)

1. Missouri Leaking Tanks
   - Who created the data: Missouri Department of Natural Resources, Air and Land Protection Division, Hazardous Waste Program, Tanks Section
   - Publication date and time: 7/15/2004
   - Publisher and place: MoDNR, Jefferson City, MO
   - Acquired from: Missouri Spatial Data information Service
   - Acquisition Date: 2006

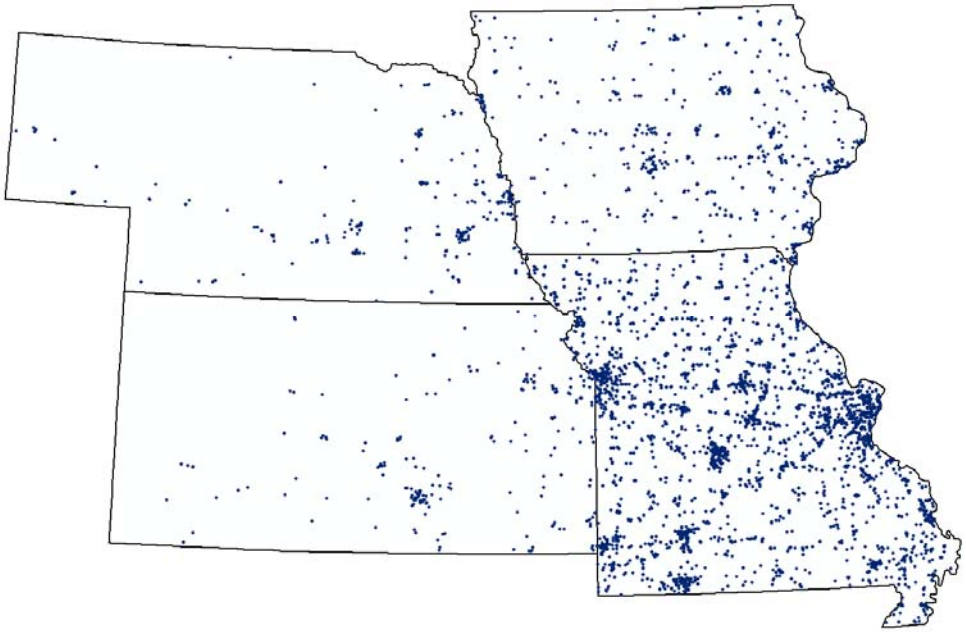
2. Iowa Leaking Tanks
   - Who created the data: Gail George, Iowa Department of Natural Resources
   - Publication date and time: 10/13/2003
   - Publisher and place: Iowa Department of Natural Resources, Iowa City, IA

- Acquired from: Iowa GIS Library
- Acquisition Date: 2006

3. Kansas Leaking Tanks
    - Who created the data: The Kansas Department of Health and Environment (KDHE)
    - Publication date and time: Unknown
    - Publisher and place: Unknown
    - Acquired from: KDHE
    - Acquisition Date: 2006

3. Nebraska Leaking Tanks (Database File)
    - Who created the data: David Chambers, Nebraska DEQ
    - Publication date and time: Unknown
    - Publisher and place: Unknown
    - Acquired from: David Chambers
    - Acquisition Date: 2006

National Pollution Discharge Elimination System (NPDES) Subset



Description

- This dataset is a subset from EPA's NPDES/PCS dataset obtained from the EPA. Since we were also classifying various other threats included in the NPDES data for this project we did not want to count the sites multiple times. It is important to note that we removed all features that had a SIC code corresponding to Waste Water Treatment Facilities (WWTF), Landfills, Temporary Permits, Mines, Confined Animal Operations (CAFOs), RCRA sites, Superfund Sites or Toxic Release sites (TRI) from the original source data. The removed datasets were quantified as separate layers.

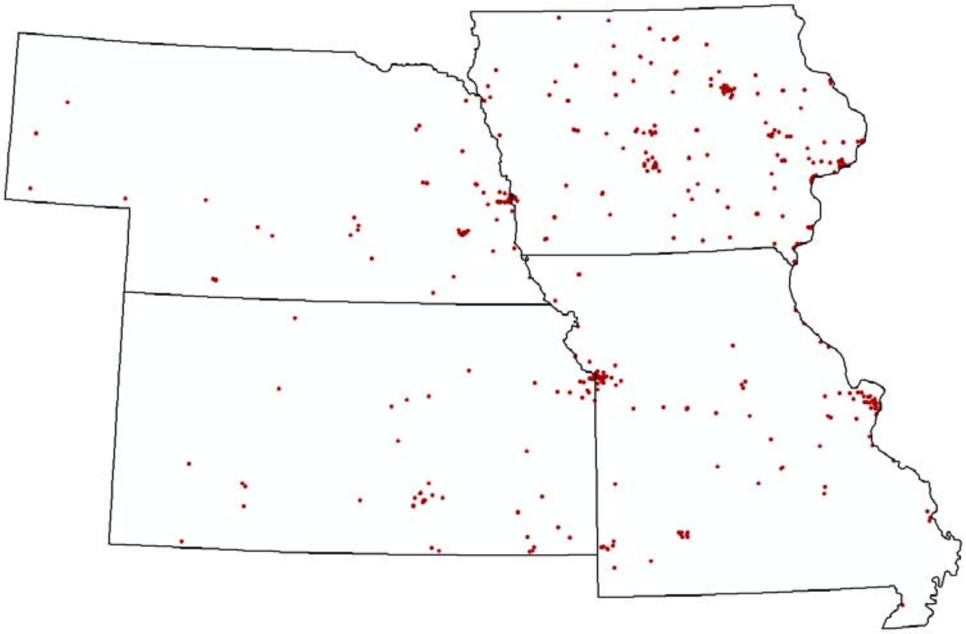
Observations

- Some points were also removed if they were time limited such as temporary building permits, these points were things such as construction sites of houses and businesses or road work.

Source(s)

1. EPA Region 7 NPDES
   - Who created the data: The Environmental Protection Agency (EPA)
   - Publication date and time: Unknown
   - Publisher and place: Unknown
   - Acquired from: EPA Region 7, Kansas City, Missouri
   - Acquisition Date: 2007

Resource Conservation Recovery Information System (RCRIS) Subset



Description

- This dataset contains all Resource Conservation Recovery Information System (RCRIS) sites in EPA Region 7 excluding those classed as Superfund or Comprehensive Environmental Response, Compensation, and Liability Information System (CERCLIS) sites.

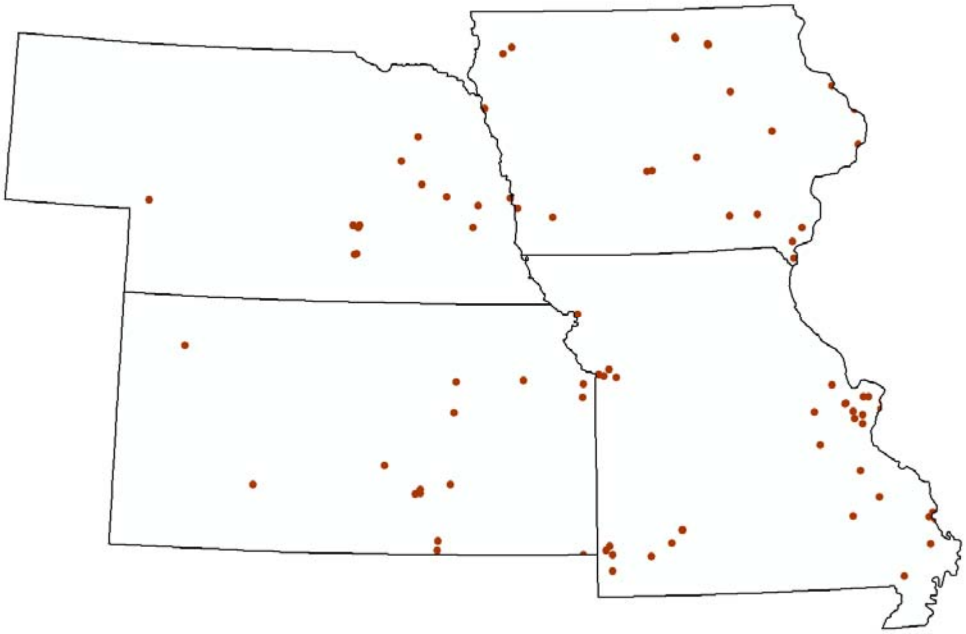
Observations

- This is not the complete RCRIS dataset. Any RCRIS sites listed as Superfund sites were removed and quantified separately.

Source(s)

1. EPA Region 7 RCRA sites
   - Who created the data: The Environmental Protection Agency (EPA) Envirofacts
   - Publication date and time: 5/2007
   - Publisher and place: Environmental Protection Agency, Headquarters, Washington, DC
   - Acquired from: EPA Envirofacts Website
   - Acquisition Date: 2007

Superfund



Description

- This dataset is a subset from EPA's Geodata dataset obtained from the EPA Envirofacts website. Features attributed as Comprehensive Environmental Response, Compensation, and Liability Information System sites (Superfund\CERCLIS) were extracted to create this shapefile.

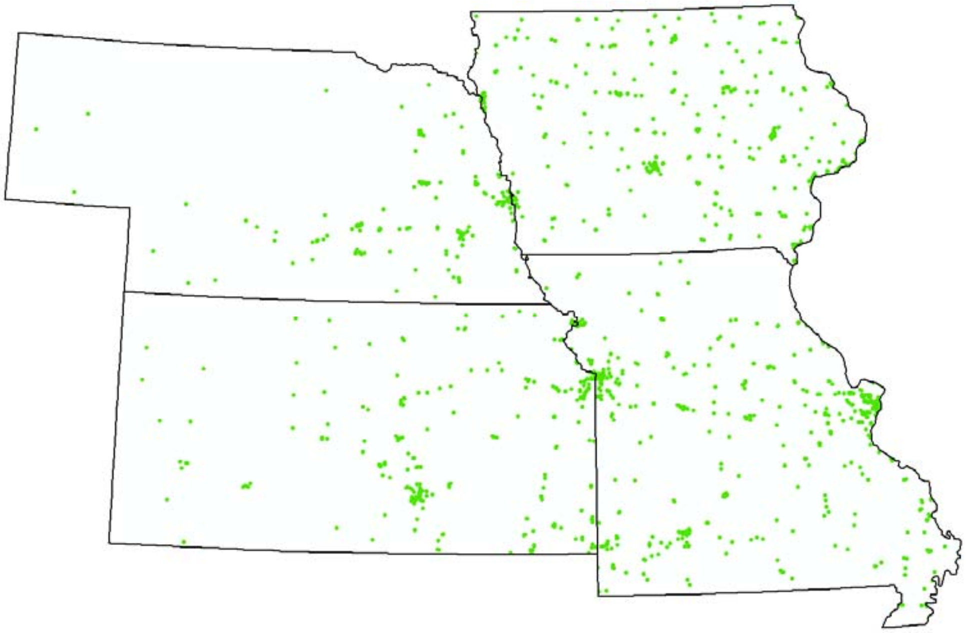
Observations

- The extraction was based on if the CERCLIS field had any attribution.

Source(s)

1. EPA Region 7 Superfund sites
   - Who created the data: The Environmental Protection Agency (EPA) Envirofacts
   - Publication date and time: 5/2007
   - Publisher and place: Environmental Protection Agency, Headquarters, Washington, DC
   - Acquired from: EPA Envirofacts Website
   - Acquisition Date: 2007

Toxic Release Inventory (TRI)



Description

- This dataset is a subset from EPA's Geodata dataset obtained from the EPA Envirofacts website. From the original dataset all features that were classified as Toxic Release sites in the TRI fields were removed for this shapefile. This dataset contains all Toxic Release sites (TRI) excepting those classed as Resource Conservation Recovery Act (RCRA) or Comprehensive Environmental Response, Compensation, and Liability Information System (CERCLIS) sites.

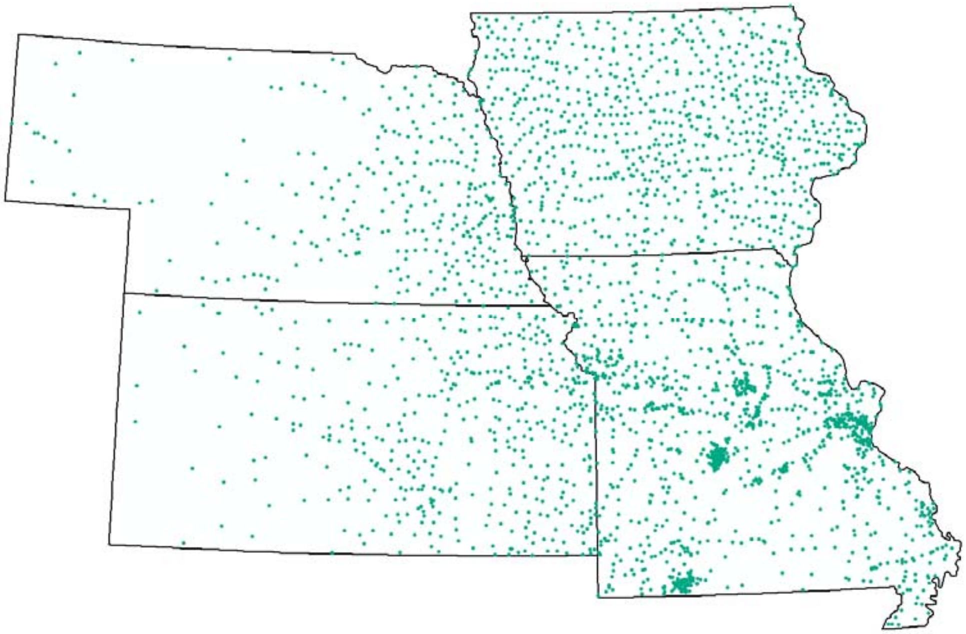
Observations

- None

Source(s)

1. EPA Region 7 TRI sites
   - Who created the data: The Environmental Protection Agency (EPA) Envirofacts
   - Publication date and time: 5/2007
   - Publisher and place: Environmental Protection Agency, Headquarters, Washington, DC
   - Acquired from: EPA Envirofacts Website
   - Acquisition Date: 2007

Waste Treatment Plants



Description

- This dataset is a subset from EPA's NPDES/PCS dataset obtained from the EPA on a data CD. All features attributed as Waste Treatment Facilities based on various SIC codes were extracted to create this data layer.

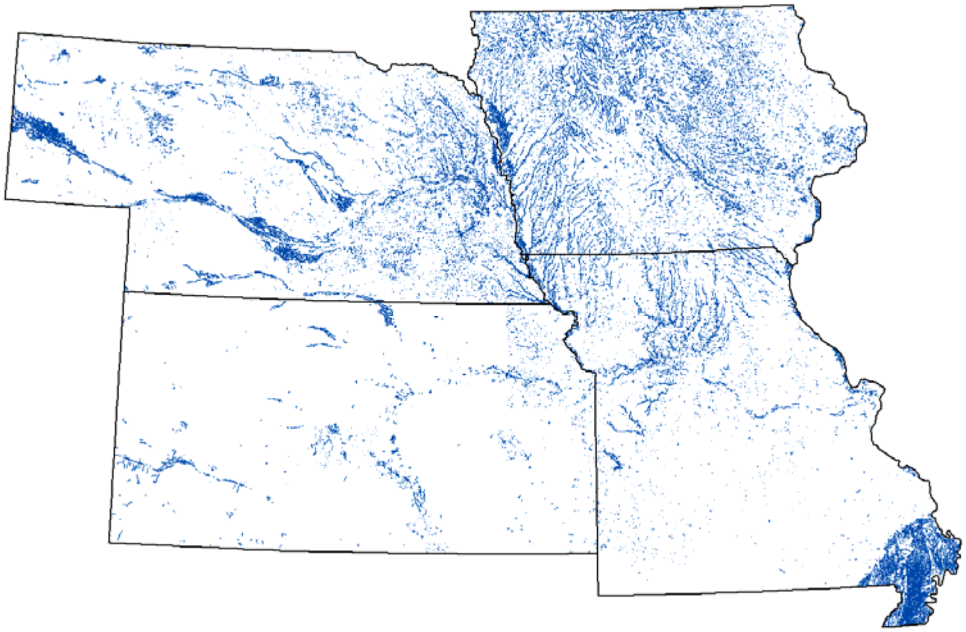
Observations

- None

Source(s)

1. EPA Region 7 NPDES
    - Who created the data: The Environmental Protection Agency (EPA)
    - Publication date and time: Unknown
    - Publisher and place: Unknown
    - Acquired from: EPA Region 7, Kansas City, Missouri
    - Acquisition Date: 2007

Channelized and Ditched Streams



Description

- This dataset consists of all ditches or channelized pieces of stream that could be identified using three input datasets; namely the 1:24,000 National Hydrography Dataset (NHD), the National Wetlands Inventory (NWI), and a modified version of the 1:100,000 National Hydrography Dataset (NHD). We used existing attribution from the aforementioned files to identify stream segments that were channelized or ditched. In addition we identified additional stream segments by visually searching for segments that appeared straightened based on professional judgment.

Observations

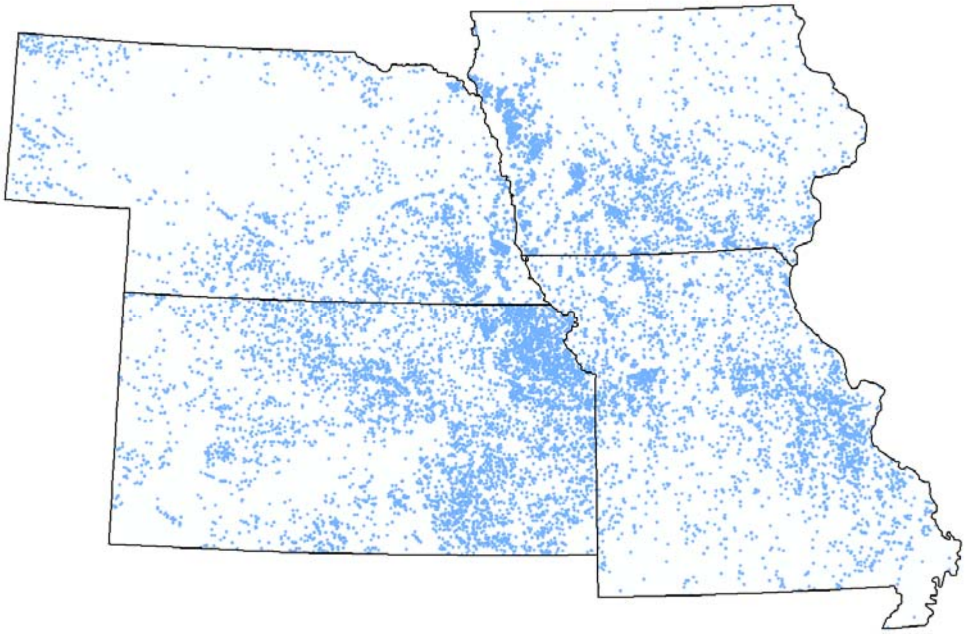
- In certain areas where we did have overlap, most of the overlap between the files was removed, however a small percentage of overlap remained because of the distance between the same features of different files.
- Some areas only have NWI data and others only have NHD data, while some areas consist of a combination of these two sources.

Source(s)

1. National Wetlands Inventory
   - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS) Wetland Mapper Team
   - Publication date and time: Unknown
   - Publisher and place: U.S. Department of the Interior United States Geological Survey (USGS) Wetlands Mapper Team
   - Acquired from: USGS Wetlands Mapper Team
   - Acquisition Date: 2007

2. 1:24,000 National Hydrography Dataset
    - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS)
    - Publication date and time: Unknown
    - Publisher and place: USGS
    - Acquired from: USGS ftp site
    - Acquisition Date: 2007

Dams



Description

- The National Inventory of Dams database contains information on 75,187 dams throughout the United States and its territories. The National Inventory of Dams is the Water Control Infrastructure, Inventory of Dams 1993-1994 report and CD-ROM. Significant changes were made to the inventory data, including the addition of new dam records and removal of breached dams, and duplicate dam records. This update was authorized under the Water Resources Development Act of 1986 (P.L. 99-662), as amended.

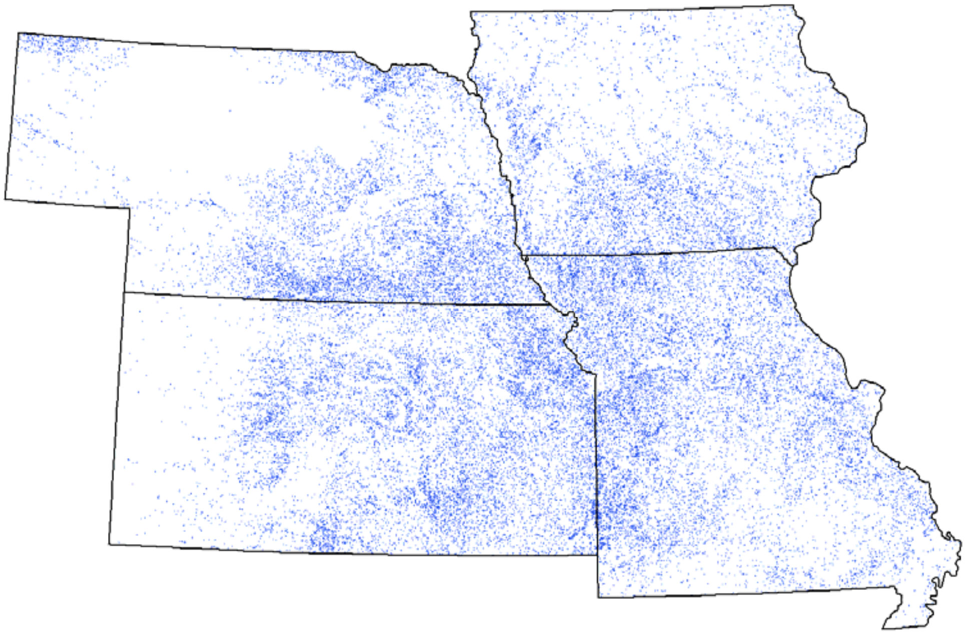
Observations

- None

Source(s)

1. EPA Region 7 Dams
   - Who created the data: U.S. Army Corps of Engineers in cooperation with FEMA's National Dam Safety Program
   - Publication date and time: 1996
   - Publisher and place: Unknown
   - Acquired from: U.S. Army Corps of Engineers
   - Acquisition Date: 2006

Headwater Impoundments



Description

- This dataset consists of all headwater impoundments that could be identified using four input datasets; namely the Elevation Derivatives for National Applications (EDNA), the National Land Cover Database (NLCD), the National Wetlands Inventory (NWI), and a modified version of the 1:100,000 National Hydrography Dataset (NHD). We used existing attribution from the aforementioned files to identify waterbodies that were impounded. We identified additional headwater impoundments by visually searching for waterbodies that appeared impounded based on professional judgment.

Observations

- This dataset represents an estimation of headwater impoundments.
- An effort was made to exclude natural waterbodies from this dataset.

Source(s)

1. National Wetlands Inventory
   - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS) Wetland Mapper Team
   - Publication date and time: Unknown
   - Publisher and place: USGS
   - Acquired from: USGS Wetland Mapper Team
   - Acquisition Date: 2007

2. 2001 NLCD
   - Who created the data: U.S. Geological Survey
   - Publication date and time: 11/13/2006
   - Publisher and place: U.S. Geological Survey, Sioux Falls, SD
   - Acquired from: MRLC Consortium - U.S. Department of the Interior United States Geological Survey (USGS)
   - Acquisition Date: 2007

3. Elevation Derivatives for National Applications (EDNA)
   - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS) EROS Data Center
   - Publication date and time: Unknown
   - Publisher and place: U.S. Geological Survey
   - Acquired from: U.S. Department of the Interior United States Geological Survey (USGS) EROS Data Center
   - Acquisition Date: 2006

4. EPA Kansas Playa Lakes
   - Who created the data: University of Kansas - Geography Department
   - Publication date and time: Unknown
   - Publisher and place: Unknown
   - Acquired from: EPA Region 7
   - Acquisition Date: 2007

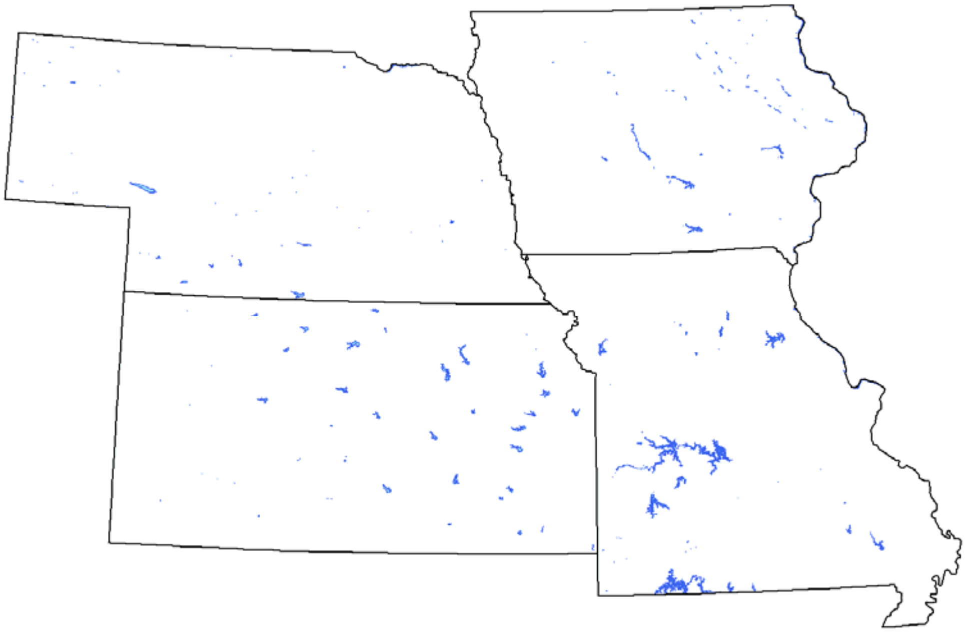
5. Iowa Designated Wetlands Setbacks
   - Who created the data: Iowa Department of Natural Resources
   - Publication date and time: 2003
   - Publisher and place: Iowa Department of Natural Resources
   - Acquired from: Iowa GIS Library
   - Acquisition Date: 2007

6. Ecoregions of the united States
   - Who created the data: United States Department of Agriculture - Forest Service ECOMAP Team
   - Publication date and time: 2005
   - Publisher and place: United States Department of Agriculture - Forest Service ECOMAP Team
   - Acquired from: Unknown
   - Acquisition Date: 2007

7. EPA Region 7 Dams
    - Who created the data: U.S. Army Corps of Engineers in cooperation with FEMA's National Dam Safety Program
    - Publication date and time: 1996
    - Publisher and place: Unknown
    - Acquired from: U.S. Army Corps of Engineers
    - Acquisition Date: 2006

8. Nebraska Dams Inventory
    - Who created the data: Nebraska Department of Natural Resources
    - National Dam Safety Program
    - Publication date and time: 1868 - present
    - Publisher and place: Nebraska Department of Natural Resources
    - Acquired from: Nebraska Department of Natural Resources
    - Acquisition Date: 2006

Major Impoundments



Description

- This dataset consists of all impoundments that intersected a Small River or larger based on stream size attribution from the modified version of the NHD stream layer used in this project. The sources of the impoundments were the1:100,000 National Hydrography Dataset Plus (NHDPlus) and the 1:24,000 National Wetlands Inventory (NWI). We used existing attribution from the aforementioned files to identify waterbodies that were impounded. We also identified additional impoundments by visually searching for waterbodies that appeared impounded based on professional judgment.

Observations

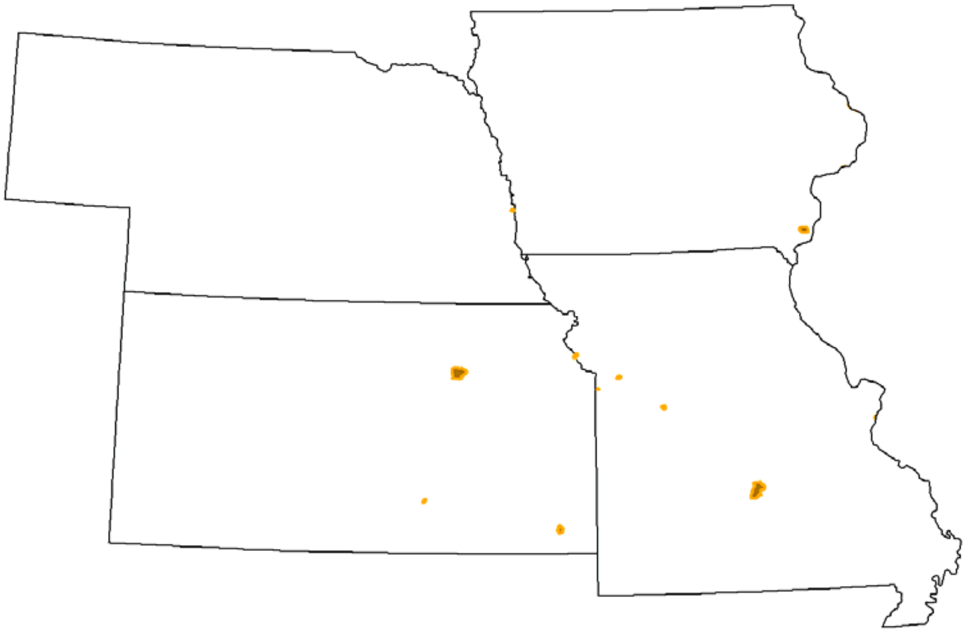
- This dataset was created using the NHD Plus and NWI to extract the water bodies for EPA Region 7.
- The impoundments were then manually compared to aerial imagery to determine if they were in fact an impoundment.

Source(s)

1. National Wetlands Inventory
    - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS) Wetland Mapper Team
    - Publication date and time: Unknown
    - Publisher and place: U.S. Department of the Interior United States Geological Survey (USGS) Wetlands Mapper Team
    - Acquired from: USGS Wetlands Mapper Team
    - Acquisition Date: 2007

2. 1:100,000 National Hydrography Dataset
    - Who created the data: U.S. Department of the Interior United States Geological Survey (USGS)
    - Publication date and time: Unknown
    - Publisher and place: USGS
    - Acquired from: USGS horizon systems website
    - Acquisition Date: 2007

Military Sites



Description

- The United States Military Installations database contains the boundaries, location, and areal information for important military installations in the United States and Puerto Rico.

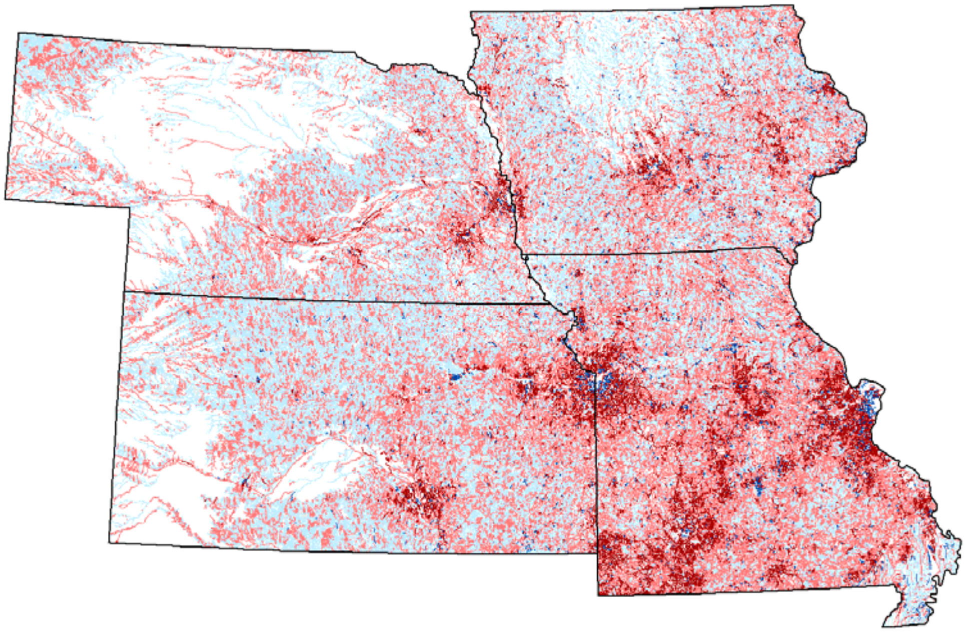
Observations

- The data is a polygon shapefile.

Source(s)

1. Military Bases
   - Who created the data: Bureau of Transportation Statistics
   - Publication date and time: 2001
   - Publisher and place: Unknown
   - Acquired from: Harvard Geospatial Library
   - Acquisition Date: 2006

Population Data



Description

- In order for others to use the information in the Census TIGER database in a geographic information system (GIS) or for other geographic applications, the Census Bureau releases to the public extracts of the database in the form of TIGER/Line files. The various population datasets contain 1990 block population data as well as 2000 block population data for each state.

Observations

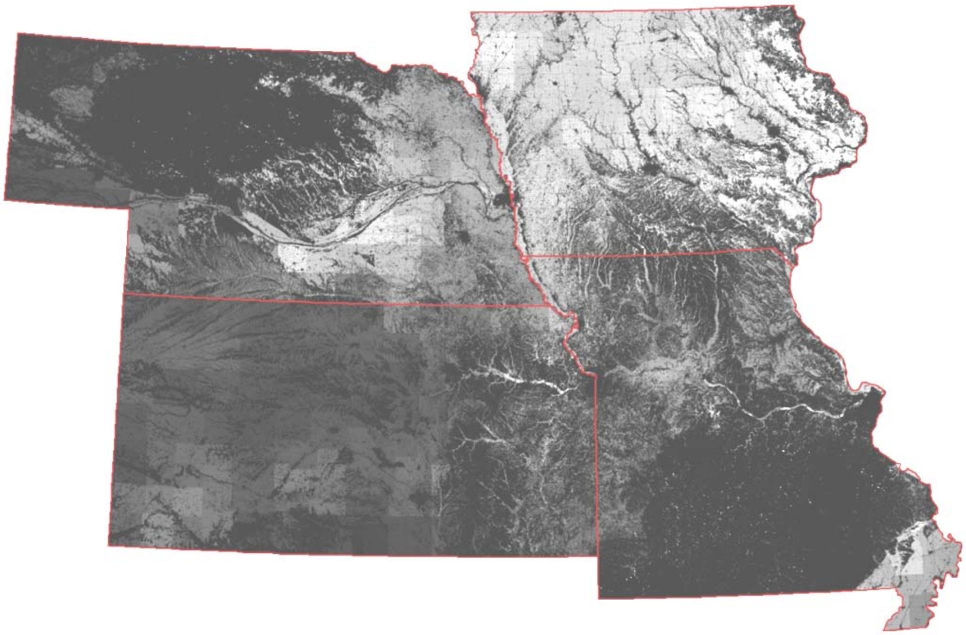
- Census block population data for both 1990 and 2000 were attached to the year 2000 census blocks to look at population changes between the two decades.

Source(s)

1. 1990 Census Block Data
   - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division
   - Publication date and time: 1993
   - Publisher and place: U.S. Department of Commerce U.S. Census Bureau Geography Division, Washington, DC
   - Acquired from: ESRI Website
   - Acquisition Date: 2007

2. 2000 Census Block Data
    - Who created the data: U.S. Department of Commerce U.S. Census Bureau Geography Division
    - Publication date and time: 2001
    - Publisher and place: U.S. Department of Commerce U.S. Census Bureau Geography Division, Washington, DC
    - Acquired from: ESRI Website
    - Acquisition Date: 2007

Estimated Crop Pesticide Grid



Description

- The goal of creating this dataset was to try and establish an estimate of cropland pesticide use over all of EPA Region7.

Observations

- This dataset represents estimated crop pesticide use based on the 1997 Agricultural Census Data and methods developed by the USGS. The grid is a 30 meter cell grid based on the 2001 National Land Cover Dataset (NLCD) and was based on the 43 most used pesticides established by the USGS.

Source(s)

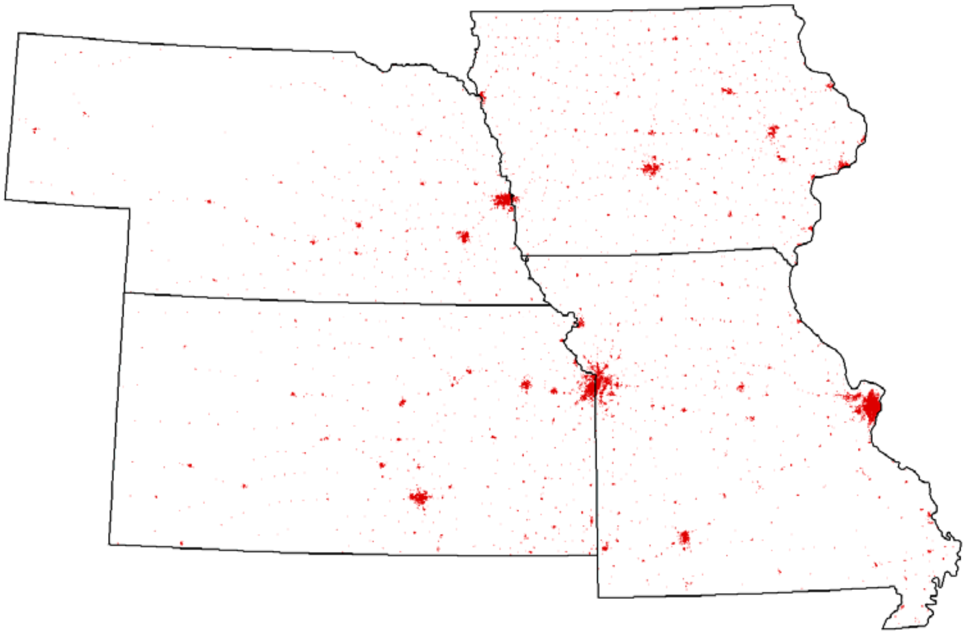
Grid created by MoRAP using the following sources:

1. 2001 NLCD
   - Who created the data: U.S. Geological Survey
   - Publication date and time: 11/13/2006
   - Publisher and place: U.S. Geological Survey, Sioux Falls, SD
   - Acquired from: MRLC Consortium - U.S. Department of the Interior United States Geological Survey (USGS)
   - Acquisition Date: 2007

2. Grids of agricultural pesticide use in the conterminous United States, 1997
   - Who created the data: U.S. Geological Survey, Naomi Nakagaki
   - Publication date and time: 1/2007
   - Publisher and place: U.S. Geological Survey, Sacramento, CA
   - Acquired from: U.S. Department of the Interior United States Geological Survey (USGS)
   - Acquisition Date: 2007

3. *Method for Estimating Pesticide Use for County Areas of the Conterminous United States*, Gail P. Thelin and Leonard P. Gianessi, 2000; U.S. Geological Survey Open-File Report 00-250

Impervious Surface



Description

- This dataset consists of all impervious surface areas excluding roads that could be identified using the 2001 National Land Cover Dataset. A 30-meter pixel often overestimates the amount of impervious surface from rural roads. As such, we decided to eliminate most roads outside of urban areas from the impervious classes in the NLCD. A shrink and expand process was used to remove roads, but maintain urban impervious.

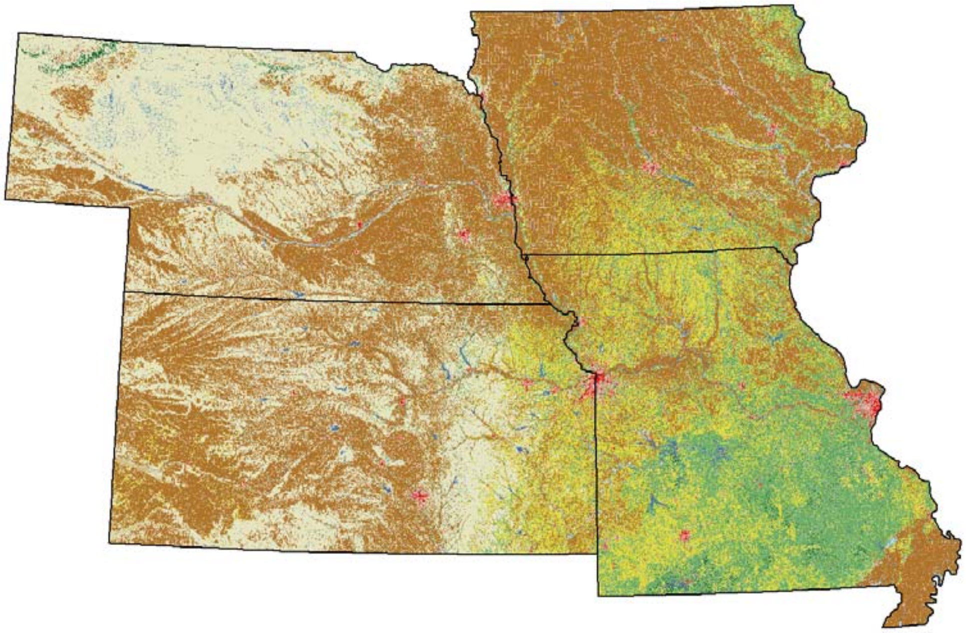
Observations

- The source data was separate from the 2001 NLCD data; it was part of the percent urban dataset.
- Due to the processing methods the depiction of impervious surface in urban areas was altered to a degree in some locations.

Source(s)

1. 2001 NLCD
   - Who created the data: U.S. Geological Survey
   - Publication date and time: 11/13/2006
   - Publisher and place: U.S. Geological Survey, Sioux Falls, SD
   - Acquired from: MRLC Consortium - U.S. Department of the Interior United States Geological Survey (USGS)
   - Acquisition Date: 2007

2001 National Landcover Dataset



Description

- The National Land Cover Database 2001 land cover layer was produced through a cooperative project conducted by the Multi-Resolution Land Characteristics (MRLC) Consortium. The MRLC Consortium is a partnership of federal agencies (www.mrlc.gov), consisting of the U.S. Geological Survey (USGS), the National Oceanic and Atmospheric Administration (NOAA), the U.S. Environmental Protection Agency (EPA), the U.S. Department of Agriculture (USDA), the U.S. Forest Service (USFS), the National Park Service (NPS), the U.S. Fish and Wildlife Service (FWS), the Bureau of Land Management (BLM) and the USDA Natural Resources Conservation Service (NRCS). One of the primary goals of the project is to generate a current, consistent, seamless, and accurate National Land Cover Database (NLCD) circa 2001 for the United States at medium spatial resolution.

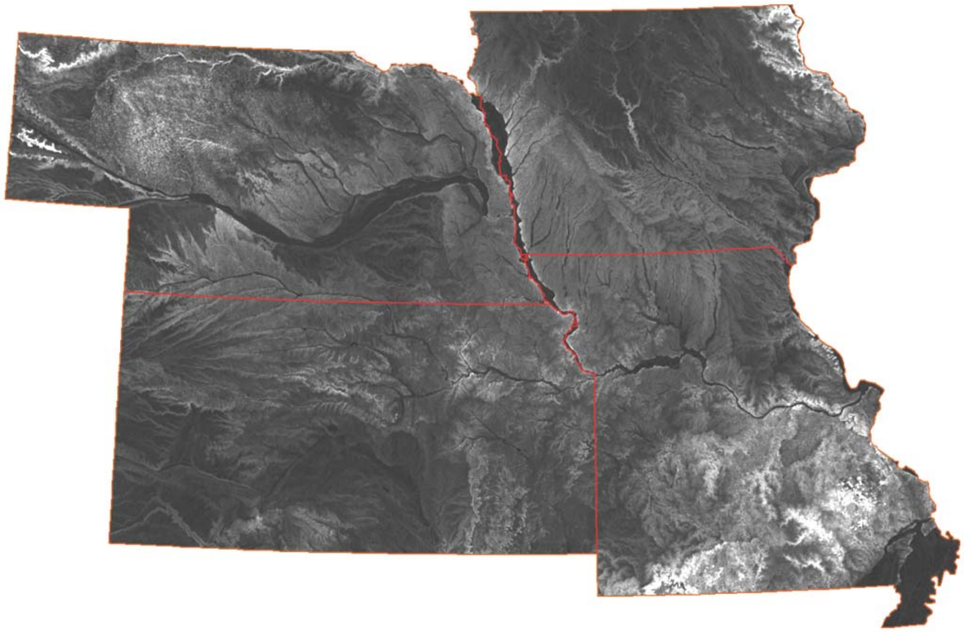
Observations

- Data was downloaded and combined into a single file for EPA Region 7.

Source(s)

1. 2001 NLCD
   - Who created the data: U.S. Geological Survey
   - Publication date and time: 11/13/2006
   - Publisher and place: U.S. Geological Survey, Sioux Falls, SD
   - Acquired from: MRLC Consortium - U.S. Department of the Interior United States Geological Survey (USGS)
   - Acquisition Date: 2007

Relief/Roughness Grid



Description

- This dataset is a grid representation of the relief or roughness of the surface of the landscape covering EPA Region 7. This dataset was created using the National Hydrography Dataset Plus (NHD Plus) DEM data.

Observations

- We used the FocalRange command in ArcMap using the Raster Calculator to make a relief grid based on the difference in elevation within a 1 kilometer diameter circle on the DEM.

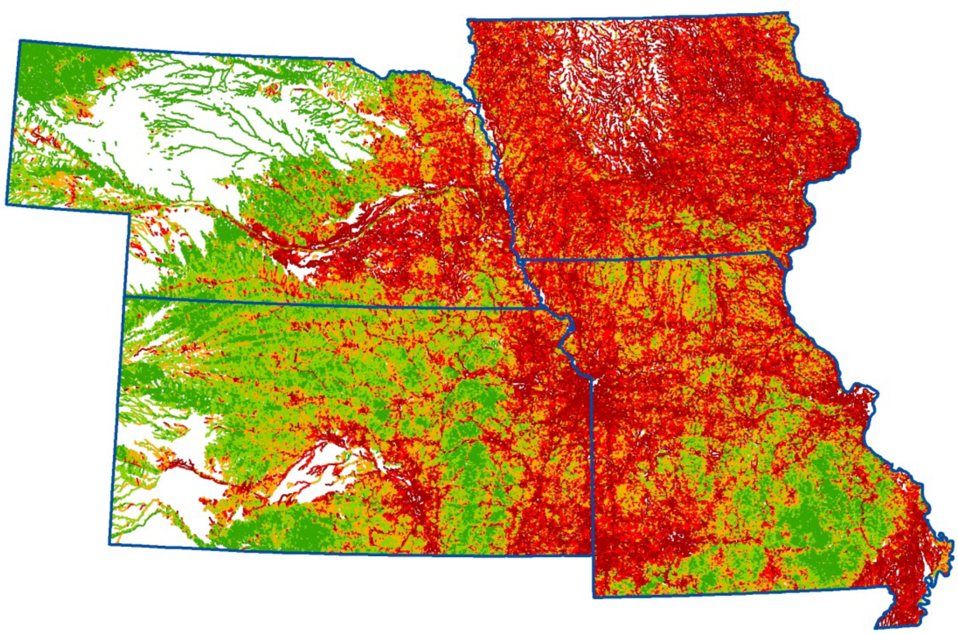
Source(s)

1. EPA Region 7 DEM's
   - Who created the data: Environmental Protection Agency & United States Geological Survey, NHD Plus
   - Publication date and time: 2006
   - Publisher and place: Unknown
   - Acquired from: NHD Plus Website
   - Acquisition Date: 2006

Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

Appendix C

Data Structure and Descriptions



Level	Folder or File Name	Description
	Data	**Folder containing all source data**
1.	**Base Data**	**Folder containing classification units**
2.	**Human Threat Data**	**Folder containing threat datasets**
3.	**Natural Data**	**Folder containing natural features data**

1.	**Base Data**	**Folder containing classification units**
	Region7_Buffers	Geodatabase containing riparian buffers
	R7_catchments.shp	Shapefile of catchment polygons of streams
	R7_Streams.shp	Shapefile of valley segment type streams

2.	**Human Threat Data**	**Folder containing various threat datasets**
	Threat Base Data	**Subfolder containing threat datasets**
	Certified Wells	**Subfolder containing state well data**
	IA Wells.shp	Shapefile of Iowa certified water wells
	KS Wells.shp	Shapefile of Kansas certified water wells
	MO Wells.shp	Shapefile of Missouri certified water wells
	NE Wells.shp	Shapefile of Nebraska certified water wells
	1990 Population.gdb	Geodatabase containing 1990 census data
	Iowa 90	Dataset of 1990 Iowa block polygons
	Kansas 90	Dataset of 1990 Kansas block polygons
	Missouri 90	Dataset of 1990 Missouri block polygons
	Nebraska 90	Dataset of 1990 Nebraska block polygons
	2000 Population.gdb	Geodatabase containing 2000 census data
	Iowa 2000	Dataset of 2000 Iowa block polygons
	Kansas 2000	Dataset of 2000 Kansas block polygons
	Missouri 2000	Dataset of 2000 Missouri block polygons
	Nebraska 2000	Dataset of 2000 Nebraska block polygons
	Airports.shp	Shapefile of airports
	CAFOs.shp	Confined animal feeding operations
	Coal Mines.shp	Shapefile of coal mines
	Crop_Pest_v1	Grid of estimated crop pesticide use
	Dams.shp	Shapefile of major dams
	EPA R7 Channelized Streams V1.shp	Shapefile of channelized or ditched streams
	EPA R7 Headwater Impoundments V1.shp	Shapefile of headwater impoundments
	Impervious	Grid of impervious surface
	Landfills.shp	Shapefile of landfills
	Lead Mines.shp	Shapefile of lead mines
	LUST.shp	Shapefile of leaking underground storage tanks
	Major impoundments.shp	Shapefile of major impoundments
	Military Bases.shp	Shapefile of military bases
	Mines.shp	Shapefile of active mines (excluding coal/lead)
	NPDES.shp	National pollution discharge elimination sites
	Oil Gas Wells.shp	Shapefile of active oil and gas wells

Level	Folder or File Name	Description
2.	**Human Threat Data**	**Folder containing various threat datasets**
	Rail Stream Crossings.shp	Shapefile of railroad stream crossings
	Railroads.shp	Shapefile of railroads
	RCRIS.shp	Resource conservation recovery act sites
	Road Stream Crossings.shp	Shapefile of road stream crossings
	Roads.shp	Shapefile of roads
	Superfund.shp	Shapefile of superfund or cerclis sites
	Toxic Releases.shp	Shapefile of toxic release sites
	WWTF.shp	Shapefile of waste water treatment facilities
	Threat_Tables	**Related tables containing threat stats**
	Distances To Threats.dbf	Database of distances from stream to threats
	Fragmentation.dbf	Database of stream fragmentation
	Human Threat Attributes.dbf	Database of stream watershed threat statistics
	Human Threat Index.dbf	Database of human threat index values

Level	Folder or File Name	Description
3.	**Natural Data**	**Folder containing natural features data**
	Natural Base Data	**Subfolder containing natural data**
	NLCD 2001	Grid of national landcover database 2001
	R7 Relief	Grid of the relief in EPA region 7
	Natural Tables	**Related tables containing natural stats**
	Landcover.dbf	Database of landcover watershed stats
	Relief.dbf	Database of relief class watershed stats
	Riparian Landcover.dbf	Database of riparian landcover stats
	Soil Hydro Group.dbf	Database of soil hydrologic group stats
	Soil Rock Depth.dbf	Database of soil depth to bedrock stats
	Soil Rock Frag.dbf	Database of soil rock fragmentation stats
	Soil Texture.dbf	Database of soil surface texture stats

Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

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Appendix D

Aquatic Survey Results: Basic Rankings of Threats to Ecological Integrity

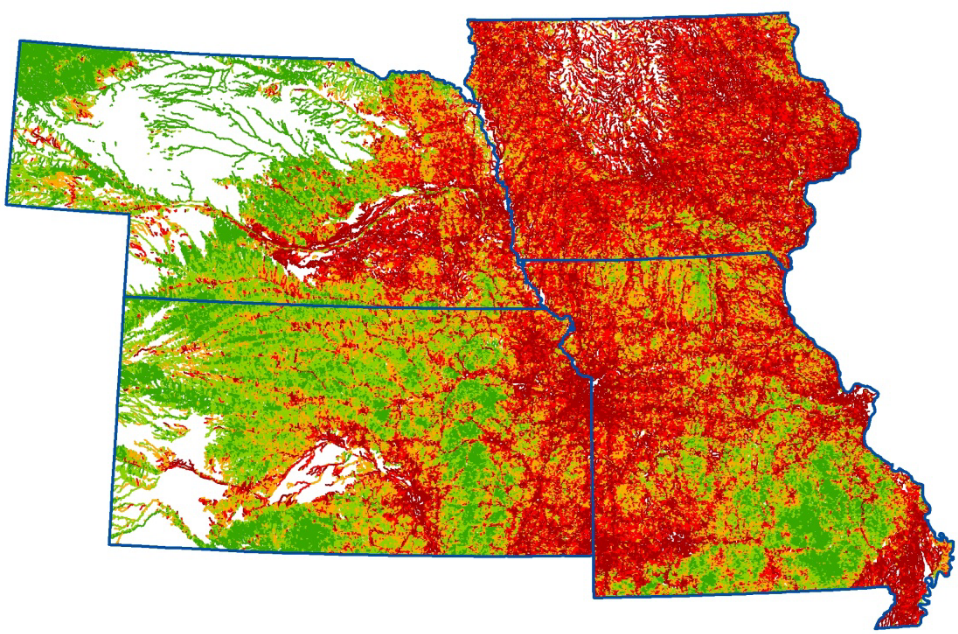

Table A.1. Survey ranking results of threat variables' impact on physical habitat.

Threats	Physical Habitat	
	Mean	Mode
Channelization	3.0	3
Instream Sand And Gravel Mines	2.9	3
Major Reservoirs	2.8	3
Navigation (Channel and Bank Maintenance)	2.8	3
Row Crop Agriculture	2.5	3
Impervious Surface	2.5	3
Headwater Impoundments (Impoundments on Second-Order and Smaller Streams)	2.5	3
Introduced Plants	2.2	2
Water Withdrawals	2.2	3
Roads (Paved And Gravel)	2.1	2,3
Flow Diversions (No Return Flow)	2.1	2
Dispersal Barriers / Low Head Dams	2.1	2
Artificial Drainage (Agricultural Field Drainage)	2.0	3
Military Sites (With Regard to Land Disturbance)	1.9	3
Flow Diversions (With Return Flow)	1.9	2
Road-Stream Crossing (Culverts And Low-Water)	1.9	1
Upland Mining	1.9	2
Ranging Livestock	1.8	2
Storm Water Systems	1.7	1,2
Bridges	1.7	1,2
CAFO	1.6	1
Golf Course	1.5	1
Point Source Discharges (NPDES: Municipal, Agricultural, And Industrial)	1.4	1
Pasture And Range Land	1.4	1
Introduced Aquatic Animals	1.3	1
Airports (National and International)	1.3	1
Waste Water Treatment Facility	1.3	1
Pipelines (Oil And Gas)	1.2	1
Landfills	1.0	1
Salt Scars (From Old Abandoned Oil And Gas Wells)	1.0	1
Toxic Releases	0.9	0
Rail Yards (Fuel And Coal Dust)	0.8	0
Recreational Use (Fishing, Canoeing, Hiking, Swimming)	0.8	1
Oil And Gas Wells	0.7	0,1
Power Lines (With Regard To Herbicide Applications)	0.7	0
Hazardous Waste Generators And Handlers	0.7	0
Underground Injection (Shallow Injections Of Waste Materials, Such As Sludge)	0.7	0
Road Salt Applications	0.6	0
Military Sites (With Regard to Chemical Contamination)	0.6	0
Septic Tanks / Lagoons	0.5	0
Former Grain Storage Facilities (Carbon Tetrachloride)	0.4	0

Key
0 = No Impact
1 = Low Impact
2 = Moderate Impact
3 = High Impact

Table A.2. Survey ranking results of threat variables' impact on water quality.

Threats	Water Quality	
	Mean	**Mode**
Waste Water Treatment Facility	2.8	3
Point Source Discharges (NPDES: Municipal, Agricultural, And Industrial)	2.7	3
CAFO	2.7	3
Row Crop Agriculture	2.7	3
Toxic Releases	2.6	3
Storm Water Systems	2.5	3
Artificial Drainage (Agricultural Field Drainage)	2.5	3
Road Salt Applications	2.4	3
Major Reservoirs	2.4	3
Upland Mining	2.3	3
Impervious Surface	2.2	2,3
Golf Course	2.1	2
Military Sites (With Regard to Chemical Contamination)	2.1	2
Flow Diversions (With Return Flow)	2.1	3
Rail Yards (Fuel And Coal Dust)	2.0	2
Headwater Impoundments (Impoundments on Second-Order and Smaller Streams)	2.0	2
Septic Tanks / Lagoons	2.0	2
Instream Sand And Gravel Mines	2.0	2
Roads (Paved And Gravel)	1.9	2
Oil And Gas Wells	1.8	2
Salt Scars (From Old Abandoned Oil And Gas Wells)	1.8	2
Airports (National and International)	1.8	2
Landfills	1.8	1
Hazardous Waste Generators And Handlers	1.8	1
Channelization	1.7	1,2
Underground Injection (Shallow Injections Of Waste Materials, Such As Sludge)	1.7	2
Ranging Livestock	1.6	2
Water Withdrawals	1.6	1
Former Grain Storage Facilities (Carbon Tetrachloride)	1.6	1
Navigation (Channel and Bank Maintenance)	1.6	2
Power Lines (With Regard To Herbicide Applications)	1.6	1,2
Pasture And Range Land	1.5	1,2
Military Sites (With Regard to Land Disturbance)	1.5	1
Flow Diversions (No Return Flow)	1.5	1
Introduced Plants	1.4	1
Pipelines (Oil And Gas)	1.4	1
Introduced Aquatic Animals	1.3	1
Dispersal Barriers / Low Head Dams	1.3	1
Road-Stream Crossing (Culverts And Low-Water)	1.1	1
Bridges	0.9	1
Recreational Use (Fishing, Canoeing, Hiking, Swimming)	0.8	1

Key
0 = No Impact
1 = Low Impact
2 = Moderate Impact
3 = High Impact

Table A.3. Survey ranking results of threat variables' impact on flow regime.

Threats	Flow Regime	
	Mean	Mode
Major Reservoirs	2.8	3
Flow Diversions (No Return Flow)	2.8	3
Impervious Surface	2.8	3
Channelization	2.8	3
Storm Water Systems	2.7	3
Water Withdrawals	2.7	3
Headwater Impoundments (Impoundments on Second-Order and Smaller Streams)	2.6	3
Artificial Drainage (Agricultural Field Drainage)	2.4	3
Navigation (Channel and Bank Maintenance)	2.2	3
Flow Diversions (With Return Flow)	2.2	2,3
Row Crop Agriculture	2.2	2,3
Roads (Paved And Gravel)	2.1	2
Waste Water Treatment Facility	1.8	2
Dispersal Barriers / Low Head Dams	1.7	1,2,3
Point Source Discharges (NPDES: Municipal, Agricultural, And Industrial)	1.7	2
Instream Sand And Gravel Mines	1.6	2
Airports (National and International)	1.6	1
Road-Stream Crossing (Culverts And Low-Water)	1.5	2,3
Upland Mining	1.4	1,2
Military Sites (With Regard to Land Disturbance)	1.3	1
Bridges	1.1	1
Golf Course	1.1	1,2
Pasture And Range Land	1.1	1
CAFO	1.0	1
Ranging Livestock	1.0	1
Landfills	0.9	1
Underground Injection (Shallow Injections Of Waste Materials, Such As Sludge)	0.7	1
Oil And Gas Wells	0.7	0,1
Introduced Plants	0.7	1
Rail Yards (Fuel And Coal Dust)	0.6	0
Salt Scars (From Old Abandoned Oil And Gas Wells)	0.6	0
Septic Tanks / Lagoons	0.5	0
Military Sites (With Regard to Chemical Contamination)	0.5	0
Power Lines (With Regard To Herbicide Applications)	0.4	0
Hazardous Waste Generators And Handlers	0.3	0
Pipelines (Oil And Gas)	0.3	0
Road Salt Applications	0.3	0
Toxic Releases	0.3	0
Recreational Use (Fishing, Canoeing, Hiking, Swimming)	0.3	0
Introduced Aquatic Animals	0.2	0
Former Grain Storage Facilities (Carbon Tetrachloride)	0.2	0

Key
0 = No Impact
1 = Low Impact
2 = Moderate Impact
3 = High Impact

Table A.4. Survey ranking results of threat variables' impact on energy and nutrient loads.

Threats	Energy / Nutrient	
	Mean	**Mode**
CAFO	2.6	3
Waste Water Treatment Facility	2.6	3
Point Source Discharges (NPDES: Municipal, Agricultural, And Industrial)	2.5	3
Row Crop Agriculture	2.4	2,3
Major Reservoirs	2.3	3
Artificial Drainage (Agricultural Field Drainage)	2.2	3
Storm Water Systems	2.0	2
Headwater Impoundments (Impoundments on Second-Order and Smaller Streams)	2.0	2
Septic Tanks / Lagoons	2.0	2
Golf Course	1.9	2
Introduced Plants	1.8	2
Channelization	1.7	3
Flow Diversions (With Return Flow)	1.7	1,2
Impervious Surface	1.6	1
Ranging Livestock	1.6	2
Introduced Aquatic Animals	1.5	2
Navigation (Channel and Bank Maintenance)	1.5	1
Pasture And Range Land	1.5	1
Flow Diversions (No Return Flow)	1.5	1
Upland Mining	1.5	1
Dispersal Barriers / Low Head Dams	1.4	1,2
Roads (Paved And Gravel)	1.4	1
Water Withdrawals	1.3	2
Instream Sand And Gravel Mines	1.3	1
Landfills	1.2	1
Toxic Releases	1.2	0
Military Sites (With Regard to Land Disturbance)	1.1	1
Military Sites (With Regard to Chemical Contamination)	1.1	1
Road-Stream Crossing (Culverts And Low-Water)	1.0	1
Road Salt Applications	1.0	0,1
Oil And Gas Wells	0.9	1
Airports (National and International)	0.9	1
Underground Injection (Shallow Injections Of Waste Materials, Such As Sludge)	0.9	1
Rail Yards (Fuel And Coal Dust)	0.9	1
Bridges	0.9	1
Hazardous Waste Generators And Handlers	0.8	0
Power Lines (With Regard To Herbicide Applications)	0.8	1
Salt Scars (From Old Abandoned Oil And Gas Wells)	0.8	1
Recreational Use (Fishing, Canoeing, Hiking, Swimming)	0.7	1
Pipelines (Oil And Gas)	0.6	0
Former Grain Storage Facilities (Carbon Tetrachloride)	0.5	0

Key
0 = No Impact
1 = Low Impact
2 = Moderate Impact
3 = High Impact

Table A.5. Survey ranking results of threat variables' impact on biotic interactions.

Threats	Biotic Interactions	
	Mean	**Mode**
Introduced Aquatic Animals	2.9	3
Major Reservoirs	2.8	3
Introduced Plants	2.6	3
Headwater Impoundments (Impoundments on Second-Order and Smaller Streams)	2.4	3
Channelization	2.3	3
Dispersal Barriers / Low Head Dams	2.3	3
Waste Water Treatment Facility	2.2	2,3
Point Source Discharges (NPDES: Municipal, Agricultural, And Industrial)	2.1	3
Row Crop Agriculture	2.0	1,3
Instream Sand And Gravel Mines	2.0	2,3
CAFO	1.9	1
Navigation (Channel and Bank Maintenance)	1.8	1
Toxic Releases	1.8	3
Water Withdrawals	1.8	2
Artificial Drainage (Agricultural Field Drainage)	1.8	1
Road-Stream Crossing (Culverts And Low-Water)	1.8	2
Impervious Surface	1.7	2
Roads (Paved And Gravel)	1.7	2
Flow Diversions (With Return Flow)	1.7	1,2
Storm Water Systems	1.6	2
Flow Diversions (No Return Flow)	1.6	2
Golf Course	1.5	1
Upland Mining	1.4	
Ranging Livestock	1.4	1
Septic Tanks / Lagoons	1.4	1
Military Sites (With Regard to Chemical Contamination)	1.4	1
Military Sites (With Regard to Land Disturbance)	1.3	1
Road Salt Applications	1.3	1
Bridges	1.3	1
Power Lines (With Regard To Herbicide Applications)	1.2	1,2
Pasture And Range Land	1.2	1
Recreational Use (Fishing, Canoeing, Hiking, Swimming)	1.2	1
Hazardous Waste Generators And Handlers	1.2	1
Airports (National and International)	1.1	1
Landfills	1.0	1
Rail Yards (Fuel And Coal Dust)	1.0	1
Former Grain Storage Facilities (Carbon Tetrachloride)	1.0	1
Pipelines (Oil And Gas)	1.0	1
Oil And Gas Wells	0.9	1
Underground Injection (Shallow Injections Of Waste Materials, Such As Sludge)	0.9	1
Salt Scars (From Old Abandoned Oil And Gas Wells)	0.9	1

Key
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Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

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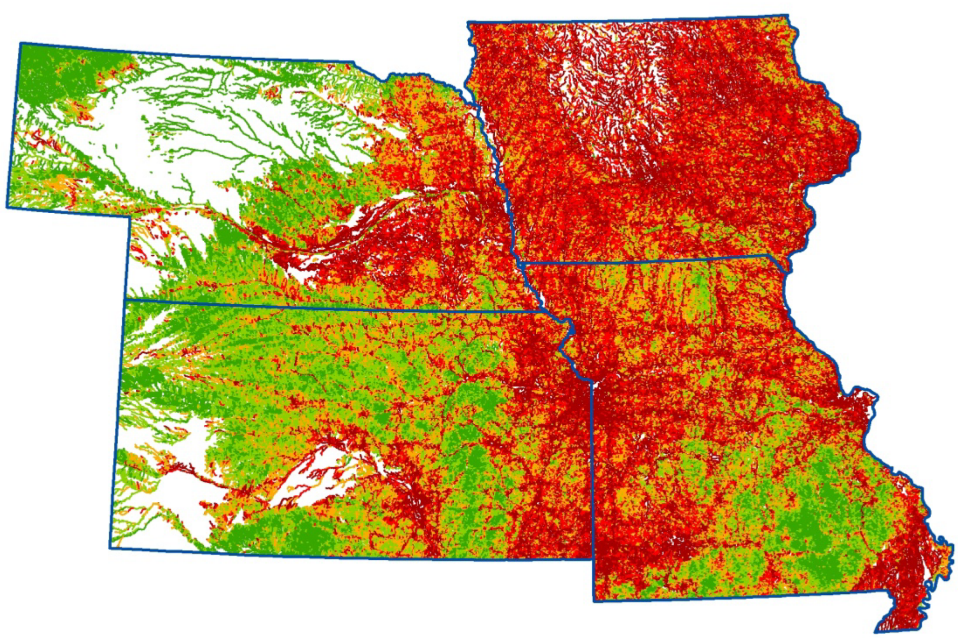
Appendix E

Literature Review of Human Threats to Aquatic Ecological Integrity



Stream Threats Literature Review

Agriculture, Concentrated Animal Feeding Operations (CAFO), and Rangeland

In North America the percentage of agricultural land within major watersheds may be as high as 66%, while the Upper Mississippi, Lower Mississippi and Missouri each contain more than 40% (Benke and Cushing 2004). A 1997 report on animal production levels in Region 7 found that Missouri had over 48 million head of livestock, followed by Iowa (>22 million), Nebraska (>10 million) and Kansas (>8 million) (http://www.scorecard.org/env-releases/aw/).

The effects of agriculture on associated rivers and streams and their biota have been thoroughly studied and documented. Studies have conclusively shown that various practices including row crop, animal production, vegetative clearing, soil compaction, water withdrawal, channelization, irrigation, and drainage cause changes in the flow regime, physical habitat, energy flow, water quality and plant and animal biota ((FISRWG) 2001; Waters 1995). Agricultural land use impacts streams by increasing inputs of nutrients, sediment and other pollutants, degrading riparian and stream channel habitat, and altering flow (Allan 2004). Major agricultural stressors are excess nutrients, sediment and pesticides which affect 55% of the impaired stream miles in the United States (Wells 1992).

Practices tend to have cascading effects. For instance, grazing or row crop production within stream riparian areas results in decreased bank stability, which leads to increased sediment yield to the channel, which causes aggradation, wider and shallower stream channels, which finally results in a loss of aquatic and riparian habitat, altered energy sources, and altered water chemistry including higher water temperatures and excess nitrogen (Allan 2004).

Flow regime

A river’s natural flow largely determines its channel morphology and the movement of sediment through the system. Variations in flows, in conjunction with other environmental cues, are the impetus for spawning, migration, and feeding for a variety of aquatic organisms. Unusual flows, either high or low, act as disturbances and are important to stream biota (Poff et al. 1997). Stream flow is affected by several factors. Independent variables such as basin shape, attitude, geology, topography, altitude, and climate, dictate the rainfall pattern—amount, timing, duration, and intensity. What happens when precipitation hits the ground depends on the quantity and type of vegetation present, and the infiltration capacity of the soil. Row crop and animal production often result in the alteration or removal of the watershed’s vegetation and soil compaction. Consequently, there may be an alteration of the stream’s water table and runoff patterns including a more “flashy” hydrograph and reductions in baseflow that turn perennial streams into intermittent or ephemeral streams ((FISRWG) 2001).

Forests increase the area from which evaporation can occur, and because trees transpire enormous amounts of water, this reduces the total volume that finds its way into stream channels through the interflow (water that infiltrates the soil and moves laterally towards the channel) or groundwater (Allan 1995). However, streams with an intact riparian buffer are narrower and deeper leading to increased water depth and higher groundwater levels, which spreads laterally to the floodplain. This means that during low flows, streams with a good riparian buffer carry more water because the vegetation increases infiltration, attenuates flows, and increases groundwater storage (Fitch and Adams 1998).

Perennial vegetation also cuts down the runoff (or "overland flows") as it tends to maintain moist permeable soils. This results in less variable stream flows (Allan 1995). Overland flows rarely occur in heavily vegetated areas, except when the soil surface is frozen or hardened by human activities, or during extreme rainfall events (Likens 1984). When these flows do occur, the result is usually a sudden increase in the amount of sediment and discharge within the receiving stream. In contrast, precipitation that penetrates a moist, undisturbed soil enters the stream slowly and is less likely to carry sediment and more likely to transport dissolved materials.

Other land management practices that impact the flow regime of rivers in agricultural landscapes include stream straightening, channelization, and artificial drainage which were widely used in the past to expand arable land, and as a short-term solution to flooding of agricultural fields (Guthrie 2000; Sovell et al. 2000; Waters 1995). Agricultural drainage lowers the water table and concentrates groundwater discharge, eliminating or fragmenting natural systems that slow and purify the runoff ((FISRWG) 2001). These modifications along with the removal of the riparian corridor, have since been shown to increase flow velocity at the previous sites of outside bends along the channel, resulting in severe bank erosion and increased sediment inputs (Darby and Thorne 1992; Williamson et al. 1992); Waters 1995), along with impaired upland or floodplain surface and subsurface flow, lower groundwater elevations, and a compressed hydrograph ((FISRWG) 2001). Other effects of these practices may include steepening of the stream's gradient that increases both current velocity and discharge leading to channel incision and loss of habitat (Gaboury et al. 1997; Harvey et al. 1985).

The consequences of an altered hydrologic regime to riverine fauna can be substantial. Historically, fish assemblages were adapted to the regular, slow springtime rises in discharge for spawning and other life activities, and were able to complete the year with moderate, but adequate flows. Many intensive land-use activities have reduced the water storage capacity of the soil, and drastically lowered water levels (Poff et al. 1997).

Physical habitat quality

The development of stream channels and entire drainage networks, and the existence of various regular patterns in the shape of channels, indicates that unaltered rivers are in dynamic equilibrium between erosion and deposition, and are governed by common hydraulic processes. A stream or river channel is a product of the water that flows along it, and is more or less adjusted to the pattern of discharge (Allan 1995). The form of a stream channel is determined and maintained by interactions between its bedload, vegetation, and valley walls, with discharges that fill but do not exceed the bank, termed "bankfull". These frequent bankfull flows maintain the existing channel form while transporting the most sediment through the system (Wolman and Miller 1960).

Agricultural practices that cause changes in the natural flow patterns (row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage) ((FISRWG) 2001) can result in channel changes that may be irreversible or require decades to recover (Wolman and Gerson 1978). A study in Iowa of the erosive effects of row crop and continuously grazed pasture versus riparian forest buffers found significantly less streambank erosion and total soil loss, and less erosion of streambank length and area for the forested area (Zaimes et al. 2006). Large increases in runoff from soil compaction or vegetative removal may result in increasing turbidity, increasing cross-sectional area of the channel, and increasing large woody debris (LWD) recruitment due to bank erosion, while causing decreases in embeddedness, cover, and LWD retention (Jacobson et al. 2001). The removal of LWD typically results in a loss of pool habitat and complexity, whereas additions of LWD increase total pool area and depth in low gradient streams (Gomi et al. 2002; Hilderbrand et al. 1997; McDonald et al. 1991). In addition, in prairie streams that tend to lack stable inorganic substrates, LWD and the associated rootmat also act as critical habitat for the invertebrate communities (Matthews 1988).

Less intense runoff events from soil compaction and vegetative removal may increase fine sediment delivery leading to simplification of stream channels (Rabeni and Smale 1995), reduced channel capacity and stability, increasing width/depth ratios and embeddedness ((FISRWG) 2001). Reductions in water levels due to water withdrawal, irrigation or drainage also lead to a loss of physical habitat by decreasing the wetted area and hence, the availability of both riffle and pool habitat, increasing habitat isolation (Power et al. 1988; Williams and Hogg 1996).

Energy sources and relations

Energy sources within a stream are derived from two basic sources: terrestrial material that is imported from the watershed and material that is fixed within the stream itself (Young and Huryn 1999). Agricultural practices which may lead to changes in a stream's energy relations include row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage ((FISRWG) 2001). The primary impacts of agricultural activities on these energy sources occur through row crop or animal production which may result in increased nutrient and sediment inputs and the loss or compositional change of the vegetation within the stream's buffer or watershed (Young and Huryn 1999).

Row crop and animal production impact the energy relations of the stream through increased inputs of nutrients and sediment. Both chemical and organic fertilizers are often applied at a higher rate than can be assimilated by crop requirements (David et al. 1997; Mallin 2000; Mallin and Cahoon 2003). Higher nutrients (in the form of fertilizers or animal manure) may increase the amount of primary production, resulting in algal blooms that reduce habitat quality and dissolved oxygen levels for aquatic inhabitants. Increased nutrients may also speed up microbial decomposition limiting its availability (Allan 2004; Cooper 1993).

Both suspended and deposited sediment may increase from runoff on plowed fields or bank failures due to livestock trampling or row crop production (Allan et al. 1997; Belsky et al. 1999; Braccia and Voshell 2006; Fitch et al. 1998). Suspended sediment may reduce light penetration resulting in reduced primary production (Ryan 1991; Young et al. 1999) or create a physical disturbance that alters or reduces the invertebrate community available as fish food (Culp et al. 1986). Deposited sediment can also interfere with macroinvertebrate feeding and respiration, or cause reduced densities via loss of preferred habitat (Quinn et al. 1997a; Rabeni et al. 2005; Zweig and Rabeni 2001).

Riparian vegetation within the stream's buffer and watershed is an important source of organic matter. A study in Maine found that mean concentrations of dissolved organic carbon were 50% higher in tributaries dominated by forest and wetland cover, as compared to those with extensive agriculture (Cronan et al. 1999). A review of studies done on 16 streams in eastern North America found that riparian deforestation not only directly reduced the amount of organic matter introduced into the stream, it also reduced and fragmented the total amount of instream habitat available for ecosystem processing of organic matter (Sweeney et al. 2004). Removal of the riparian canopy increases the amount of sunlight and resultant primary production changing the ratio of autochthonous to allochthonous inputs to the stream (Cummins 1974). In addition, riparian canopy cover moderates water temperature that affects the rate of organic matter processing (Pinto et al. 2006).

Energy relations within prairie streams differ from streams that occur within areas that originally were forested. Because their canopies are often open, with high insolation, and low litter inputs [grasses composed 57% of the litter in Kansas streams (Gurtz et al. 1982)], primary production by benthic algae produces most of the organic matter for these systems resulting in autotrophic systems except perhaps when turbidities are high (Zale et al. 1989). Studies of primary production within prairie streams of Illinois revealed that these streams tended to be more productive than streams that drain forests or more arid watersheds. The increased productivity was linked to shallow, warm, open-canopied headwaters with low turbidity (Wiley et al. 1990). This suggests that agricultural practices that increase the supply of fine sediment to streams such as row crop and animal production may reduce the levels of primary production, leading to a subsequent reduction in secondary production. Another factor that may impact primary production in these streams is runoff containing herbicides such as atrazine which may inhibit the growth of algae, phytoplankton and macrophytes (Solomon et al. 1996).

In both forested and prairie streams, litter processing is regulated by flow (Gurtz et al. 1982; (Blood et al. 1986; Hutchens and Wallace 2002)). Changes in the natural flow regime caused by row crop and animal production may impact litter processing in the affected streams. Changes in the delivery of fine sediments or flow regime would be expected to alter the food web of the affected stream reach (Vannote et al. 1980).

Water quality

Survey work done 200 years ago indicates that some higher order prairie streams were naturally turbid (Matthews 1988). However, for most prairie streams turbidity has increased since the native sod was turned for agricultural purposes. In presettlement times, the small prairie streams of southern Iowa were described as follows by Herbert Quick (Quick 1925): "Between the watersheds and at distances of two or three miles from one another were little clear brooks with banks of black sod, their waters flowing on floors of bright colored glacial pebbles; their expansion little pools covered with the pads of the yellow pond lily or lotus."

Agricultural landscapes are more sensitive to climatic variability than natural landscapes because tillage and grazing typically reduce water infiltration and increase rates and magnitudes of surface runoff resulting in increasing sediment and nutrient loads (Knox 2001). Sediment has been documented as the number one cause of impairment of streams in the United States (USEPA 2003), with about 40% of assessed river miles affected (USEPA 2000). Erosion from row crops is a more important source in the Midwest and Southeast, while livestock grazing is more important in the West (Waters 1995). However, the greatest overall source of sediment to streams is row crops and other types of field cultivation (Robinson 1971). Barren land in Missouri has been reported to lose soil at a rate 123 times that of similar land that is covered in sod (Pimentel et al. 1995).

Work done in northern Missouri on 15 streams within the Upper Salt River basin reported a significant positive relation between total suspended solids (TSS) and the percentage row crop within a watershed, and a negative correlation between TSS and the percentage of forest (Skadeland 1992). A study of eight streams within watersheds of the Ozark Border looked at the effects of percent forest, pasture and row crop on TSS. The results showed that TSS was significantly negatively correlated with percent forest and positively correlated with percent pasture and row crop (Lohman 1988). Regression coefficients revealed that at low flow, the percent of pasture had a much greater effect than that of row crops on TSS, but at moderate flow the regression coefficients were the same. A follow-up study of suspended sediment concentrations at low flows within the Ozark Border, and Ozark Highlands showed that row crop agriculture had the strongest relations to TSS (Perkins et al. 1998). The mixed results of these studies highlight the difficulties of apportioning responsibility in large watersheds with multiple land uses.

Riparian buffers have been shown to reduce the amounts of sediment and nutrients entering streams and rivers. A modeling study in South Carolina found that land use within 150 m of a river was a better predictor of nutrient concentrations than land outside that area (Tufford et al. 1998). In studies in Minnesota where best management practices were compared to conventional agricultural practices, percent fines were negatively correlated with riparian buffer width (Nerbonne and Vondracek 2001, Zimmerman et al. 2003). In Alabama, depending on soil characteristics, vegetation, and slope, various riparian buffer widths were found to assimilate or detain more than 90% of the nitrate passing through the buffers (Basnyat et al. 2000). A study of thin soils within the Inner Coastal Plain and Piedmont watersheds of the Chesapeake Bay reported that riparian forest buffers improved water quality by retaining 50%-90% of the total loading of nitrate in shallow groundwater, sediment in surface runoff, and total N in both surface runoff and groundwater, while retention of phosphorus is generally much less (Lowrance et al. 1997).

A study in Michigan found that major watersheds dominated by row crop agriculture had the highest alkalinity, total dissolved solids and nitrate + nitrite concentrations (Johnson et al. 1997). Studies in karst basins of West Virginia found a strong linear relationship between nitrate concentration and percent agricultural land (Boyer and Pasquarell 1995), while a study of the effects of forested, agricultural and urban areas on water quality and aquatic biota in the Piedmont ecoregion of North Carolina found that agricultural lands produced the highest nutrient concentrations (Lenat and Crawford 1994). Long-term nitrogen data from streams within the

Konza prairie showed a gradient of increasing NO3- concentrations from the pristine upland stream reaches to the more agriculturally-influenced lowland reaches. Minimum NO3- concentrations in the agriculturally influenced reaches were greater than at any time in prairie reaches (Kemp and Dodds 2001). Chemicals such as carbon tetrachloride released to the soil may adsorb to organic matter and be transported into water resources where evaporation is considered to be the main process for the removal of carbon tetrachloride from aquatic systems (Organization 1999).

In eastern Iowa the results of analyses of row crop and CAFO densities showed that they were significantly and independently correlated to higher nitrate concentration in streams with correlation coefficients of 0.59 and 0.89 reported between nitrate concentration and row crop and CAFO density (Weldon and Hornbuckle 2006). In areas with high animal concentrations, inputs of animal waste can increase the nutrient loads above standards and introduce bacteria and pathogens (Collins et al. 2005; (FISRWG) 2001). This was shown in West Virginia where the karst resurgence springs in areas of heavy cattle grazing (79%) were contaminated with fecal bacteria (Pasquarell and Boyer 1995). Another problem associated with CAFOs is the practice of spreading manure for fertilizing fields. When applying N at or near the nutrient uptake rate, P may be applied at up to 3 times the uptake rate. This excess P has been shown to leach off very rapidly and end up in streams (McFarland and Hauck 1999). In North Carolina, a study of seepage from an anaerobic swine wastewater lagoon found that associated surface waters were impacted dramatically with NO3-N and NH4-N downstream from the seepage plume averaged 1.5 and 12.7 mg/L, respectively (Dukes and Evans 2006).

Another common source of nutrients to surface waters besides row crop and animal production is through subsurface drainage. Watersheds in the Midwest are mostly drained by tile drainage systems (Hatfield et al. 1998). Although tile-outlet terraces in Nebraska have been shown to effectively reduce sediment and nutrient concentrations in runoff (Schepers et al. 1985), this method of drainage may provide a rapid conduit for leached nitrate–nitrogen to move rapidly into surface water supplies (Zucker and Brown 1998). Nutrient loads and availability are also impacted by the sediment load via transport or absorption (USEPA 1986).

Oxygen is often nearly absent in groundwater because of the microbial respiration that occurs from organic matter processing during its slow movement through the soil. Once groundwater enters the stream, oxygen enters the water largely through diffusion from the air. Rates of diffusion are determined by factors such as sediment type, gradient, and current velocity all of which may be changed by agricultural practices. Other factors that influence the amount of oxygen in the water include water temperature, photosynthesis, groundwater inputs, organic pollution and turbidity. Agricultural practices which may lead to changes in oxygen levels include row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage ((FISRWG) 2001).

A river's temperature regime is determined by several factors including its insolation, hydrology, and channel morphology (Hawkins et al. 1997). All of these factors may be impacted by agricultural practices that cause changes in water depth, turbidity, soil compaction or riparian vegetation (Davies and Nelson 1994; Davies-Colley and Quinn 1998; Hetrick et al. 1998; Hill et al. 1995). In small streams, the deeper the water the less is the daily variation, which is caused primarily by radiation into the water. High turbidities resulting from bank erosion or runoff over compacted soil may increase water temperatures by absorption of radiation, or decrease water temperatures by back-radiation (USEPA 1986). Agricultural practices which may lead to changes in oxygen levels include row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage ((FISRWG) 2001). The amount of light reaching a stream depends on the time of year, geography, altitude, weather, and local factors such as water depth and clarity. Light in the water column may be used for photosynthesis, absorbed and converted to heat (more than half the light entering the water), or scattered. Increased turbidity of the water reduces penetration of light and intensity of light in the water (Davies-Colley et al. 1992). Light affects water temperature that in turn exerts a major influence on thermal behavior of the biota and primary production rates. Agricultural practices which may lead to changes in oxygen levels include row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage ((FISRWG) 2001).

Biotic Interactions

Any human activity that disrupts or intensifies the transfer of water or other materials such as organic matter, nutrients, sediments, or contaminants from terrestrial areas to freshwaters, or any activity that simplifies the landscape adjacent to water bodies, is expected to have wide-ranging effects on the affected stream and its inhabitants (Palmer et al. 2000). Land use change is expected to have a dramatic global effect on biodiversity over the next 100 years, especially for freshwater ecosystems. Stream biodiversity is predicted to be affected most strongly by land use, followed by biotic exchange (Sala et al. 2000). A survey of the problems affecting 135 imperiled freshwater species of fishes, crayfishes, dragonflies and damselflies, mussels, and amphibians in the U.S. indicated that altered sediment loads and agricultural nonpoint pollution were a leading threat (Richter et al. 1997).

Agricultural practices that may impact the inhabitants of streams and rivers either directly or indirectly include row crop and animal production, vegetative clearing, soil compaction, water withdrawal, channelization, and irrigation and drainage ((FISRWG) 2001). Increases in sediment load from grazing or row crops may result in biological changes within invertebrate and fish communities (Quinn et al. 1997b). Changes in the aquatic biota have also been associated with the removal of riparian vegetation both within the immediate buffer area (Davies et al. 1994; Whiles et al. 2000) and within the watershed as a whole (Kedzierski and Smock 2001; Stewart et al. 2001; Stone and Wallace 1998). Animal production such as poultry and swine facilities and cattle grazing adds nutrients and bacteria to streams (Fisher et al. 2000; Sheffield et al. 1997; Wernick et al. 1998), while row-crop production adds nutrients and pesticides (Cuffney et al. 2000). It is a long-held belief that nutrients and pesticides have detrimental effects on aquatic biota (Hilsenhoff 1977). Recent studies have substantiated clear connections between increased nutrients and pesticides with changes in biological measures of algae, invertebrates, and fish (Cuffney et al. 2000; Pan et al. 1999; Shieh et al. 1999, Gormley et al. 2005). Carbon tetrachloride associated with abandoned grain storage facilities may be introduced into surface and ground waters. It appears to be a low risk to adult aquatic organisms. However, it may present a risk to embryo and larval stages at or near contaminated sites. Studies have shown that it has a moderate potential for bioaccumulation under conditions of constant exposure, but the compounds short tissue lifetime reduces this tendency (Organization 1999).

Changes in flow can eliminate taxa. High flows that occur at critical times in macroinvertebrate or fish life cycles may displace some species or disrupt reproduction, while low base flows may result in isolation or elimination of critical pool habitat (Allan 2004). Changes in channel structure associated with soil compaction, water withdrawal, channelization, and irrigation and drainage would also be expected to impact plant, invertebrate and fish assemblages ((FISRWG) 2001).

Macrophyte production in mid-sized rivers appears to be most affected by current velocity and light (Westlake 1975) factors impacted by many agricultural practices. Nutrients are not usually limiting factors in large rivers (Allen 1995). In studies of phytoplankton in the Kentucky River, density and composition of phytoplankton were positively correlated to agricultural land use, while diversity decreased with nutrient enrichment (Stevenson and White 1995).

The conversion of forestland to pasture can have a major impact on aquatic invertebrates of adjunct streams (Sweeney 1995). The resultant sedimentation is a major stressor to macroinvertebrate fauna (Braccia et al. 2006), while other stream changes, including channel narrowing, increasing water velocities, light penetration, water temperatures, and periphyton biomass along with decreases in leaf litter and woody debris input, were found to result in densities of mayflies, stoneflies and caddisflies that were 2 – 3 times higher in forested streams versus pasture (Quinn et al. 1997). An examination of the impact of riparian buffer on macroinvertebrates in Ontario revealed that loss of the riparian buffer resulted in lower of EPT richness and Simpson diversity (Rios and Bailey 2006). Nutrient additions have been shown to change the invertebrate community in Ontario streams (Corkum 1996), while large influxes of nitrogen, which appears to be the limiting factor in Ozark streams (Lohman et al. 1991), may result in significant changes in the affected invertebrate community.

Studies of agriculturally impacted streams in Colorado showed that those streams had comparable macroinvertebrate densities to reference streams but lower richness and relative abundance of scrapers and shredders (Shieh et al. 1999). Disturbance by grazing cattle has been correlated with more chironomid larvae (del Rosario et al. 2002) and fewer mayflies (Reed 2003) and other changes in the invertebrate community (Rothrock et al. 1998; Scrimgeour and Kendall 2003; Weigel et al. 2000). Other studies have shown increased suspended sediment causes changes in species composition and reduced abundance in zooplankton (Kirk 1991a; Kirk 1991b; Kirk 1992; Kirk and Gilbert 1990), and changes in species composition, reduced abundance (Gislason et al. 2001; Haynes 1999; Lloyd et al. 1987), increased drift (Culp et al. 1986; Doeg and Milledge 1991; Runde 1999), and impaired feeding of macroinvertebrate communities (Aldridge et al. 1987).

Matthews (1988) discussed Midwest prairie streams prior to settlement and suggested that small streams have probably undergone more change than the larger rivers, with many that were clear 100 years ago now being highly turbid. Larimore and Bayley (1996) reported on the presettlement conditions of prairie streams and their fish communities in Champaign County, Illinois. Before settlement, marshy swales acted as the headwaters of this flat, diffuse prairie. As agriculture moved into these areas, the swales were drained and native sedges and grasses were replaced with corn and wheat. They reported a loss of 18 species of fishes from these streams since the first survey done in the 1890's. Karr el al. (1985) provided another account of the changes since 1850 to headwater prairie fish communities of the Illinois River system in Illinois, and the Maumee River system in Ohio and Indiana. Comparisons between early collections and subsequent ones indicated that headwater specialists are especially susceptible to extirpation. Six headwater species that used clear water and clean gravel, or marshy areas (destroyed by draining) have been extirpated in the Maumee River, while four have been extirpated from the Illinois headwaters. Populations of invertivores and herbivores in headwaters of both rivers have also declined dramatically during this time. In addition, destruction of spawning habitat in headwater areas was reported as the critical factor in the declines of many mid-river species.

In a study of fish community samples collected at 22 stream reaches within the Ozark Plateau (Petersen 1998), differences in the fish communities were attributed to differences in land use and related water quality and habitat characteristics. Communities from agricultural reaches tended to have more species, increased relative abundance of stonerollers and members of the sucker family, and decreased relative abundance of members of the sunfish and darter families. In addition, higher levels of turbidity or suspended solids generally has been found to have a negative influence on the number of sensitive species, number of individuals, proportion of lithophilic spawners, and proportion of omnivores (Karr et al. 1986; Schleiger 2000, Waite and Carpenter 2000). Other studies of the effects of land use on fish communities have shown that the proportions of tolerant species and proportions of species with deformities, eroded fins, lesions, or tumors were positively related to increased levels of turbidity or suspended solids.

Studies in the Midwest have reported strong positive relations between a biotic index for fish communities and the % forested watershed (Roth et al. 1996; Wang et al. 1997), while overgrazing has been reported to cause reductions in fish species richness (Fitch et al. 1998). In North Carolina, a study of the impacts of land use on the fish assemblage found that the primary gradient affecting sites in this watershed was related to high levels of agricultural land cover, nitrate plus nitrite, sulfate, specific conductance, and sediment. Changes in the fish assemblage were represented as a trophic shift from specialized insectivores to generalized insectivores and an herbivore (Rashleigh 2004).

Thresholds

Reported thresholds for negative effects of agriculture have been reported in the literature (Allan 2004). Conversion from native forest to improved pasture in New Zealand for greater than 30% of a watershed has been reported to result in a decrease in total richness, EPT richness and diversity of the invertebrate assemblage (Quinn and Hickey 1990). Fish communities of streams in Wisconsin have been negatively impacted when agricultural land use exceeded 50% of the watershed (Wang et al 1997) or 10% of the buffer (Fitzpatrick et al. 2001). However, a study of multiple sites throughout the U.S. showed that fish communities within agricultural streams were more likely to be impacted by the water quality than a particular threshold of land use (Meador and Goldstein 2003).

Water Management including large dams, reservoirs and dispersal barriers

The effects of large impoundments on rivers and streams and their biota have been thoroughly studied and documented. Studies have conclusively shown that large impoundments cause changes in the flow regime, physical habitat, energy flow, water quality (Baxter 1977; Petts 1980a; Richard 2001), and the plant and animal biota (Baxter 1977; Burkhead et al. 1997; Bednarek 2001; Poff and Hart 2002). Graf (2006) analyzed upstream and downstream reaches of 36 American rivers with large dams and found that on average they reduced annual peak discharges 67%, decreased the ratio of annual maximum/mean flow 60%, and could alter the timing of high and low flows by as much as half a year. When compared to similar unregulated reaches, regulated reaches had 32% larger low flow channels, 50% smaller high flow channels, 79% less active flood plain area, and 3.6 times more inactive flood plain area. Large dams may include those with hypolimnetic or overflow release, and low head dams.

Unless there are special modifications all large dams act as dispersal barriers. Many of the effects on the environment and the aquatic inhabitants are related to the size and operational mode of the dam (Poff and Hart 2002). Dam height and width influences many aspects of the environment created by its construction. These include factors such as temperature stratification and thermal regime modification, effectiveness as a barrier to both biotic migration and sediment transport, and ability to store peak flows. The operational mode or hydraulic residence time (HRT) of the dam also plays a large role in the consequent biophysical regime. HRT is the ratio of the storage volume of the reservoir to its flow-through rate. This key factor can influence flow regime, sedimentation, type and rate of biogeochemical cycling, development of planktonic communities, transport of biota through the reservoir to downstream reaches, and thermal stratification (Poff and Hart 2002). Reservoirs with a residence time of less than 30-40 days tend to have river-like ecological structure and function, in contrast to reservoirs with a residence time >100 days that may be more similar to natural lakes (Soballe et al. 1992).

Reservoirs also aid the spread of alien species by increasing the abundance of standing freshwater habitats and acting as stepping stones for their dispersal through increased interconnectivity of contributing rivers and streams. In addition, reservoirs may be more conducive to inhabitation by alien species than natural lakes due to their interconnectivity, physiochemical properties and higher levels of disturbance (Havel et al. 2005).

Flow regime

The impoundment of a large river produces an obvious change in the flow regime. Generally, there is a reduced overall discharge, substantially truncated peak flows, higher low flows, an increase in near bankfull flow durations, a decrease in flow variability, and a shift in the timing of the natural hydrograph (Studley 1996; Vorosmarty and Sahagian 2000; Magilligan and Nislow 2005; Poff et al. 2006). The reduction in discharge means a reduction in river power, which directly affects the competence and capacity of the river. The results of less power are aggradation upstream since the river no longer has the power to move large rocks, and the settling of fine materials into the reservoir resulting in less turbid waters downstream. However, the sediments that are withheld (up to 95% of the total load) may limit the river downstream. Clear-water releases can cause downstream erosion as the river attempts to regain sediment equilibrium. The channel may become coarse or armored, riffle-pool sequences are lost, and the stream bank collapses resulting in riparian losses. Channel incision and a lowered streambed may also occur (Kondolf 1997). This can lead to lower groundwater tables and affect riparian zones by limiting access to water (Petts 1984; Bednarek 2001; Duncan 2002).

The truncation of peak flow results in the loss of several of the beneficial effects of flooding such as habitat formation of scour pools, riffles, and complex wood accumulations; the inundation of floodplains that act as nurseries, feeding grounds and refugia; the transportation of food into the river from the terrestrial ecosystem; and the cleansing and resorting of gravels and fine sediments necessary for the biota (Gregory et al. 2002; Magilligan et al. 2003). Higher low flows may sound beneficial, but the impacts on native biota that have adapted to reduced flows, may be devastating (Trotzky and Gregory 1974; Baxter 1977; Dynesius and Nilsson 1994). In a review of jeopardized species of fishes in the southeastern US, Etnier (1997) listed over 40 lotic species that were impacted by altered flows, and a similar nationwide study reported that altered hydrologic regimes due to dams and impoundments were one of the leading threats to aquatic species in the United States (Richter et al. 1997).

Finally, the overall water budget for the drainage basin is altered as a result of the trapping or rapid export of freshwater runoff, the subsequent loss of surface runoff to groundwater supplies, and the evaporation of large quantities of stored water from the reservoir (Vorosmarty and Sahagian 2000). Evaporative losses depend primarily upon the mean surface area of the reservoir and the potential evaporation. Reported losses are greatest from reservoirs in the southern Great Plains where annual evaporation is greater than annual rainfall (Petts 1984). In large hydroelectric facilities the peaking effects necessary for energy generation can result in reduced biotic productivity related either directly to the continual flow variations or indirectly due to related changes in water depth, temperature or scouring of sediments (Cushman 1985), while dredging for navigation may alter the distribution of scour, possibly increasing the depth of scour at some locations within the navigation channel (Heath et al. 1999).

Physical habitat quality

One of the most obvious effects of large dams on rivers is the change from a lotic to lentic environment. However, large impoundments are hybrid environments that lack the productive littoral zone of natural lakes and the heterogeneity of riffle/run/pool habitats of rivers and streams (Burkhead et al. 1997). Reservoirs regulate flow and obstruct the downstream transmission of sediment loads so that downstream reaches experience a reduction in the magnitude and frequency of discharge (Ward and Stanford 1995; Magilligan et al. 2003). Downstream geomorphic and ecological impacts of a dam vary with the structure and operational strategy, the preexisting characteristics of the downstream river channel such as local geomorphic constraints, and sediment supply. Riverbeds may incise or aggrade, affecting floodplain inundation; and bed sediments may become coarser or finer. Downstream incision results because large dams can trap virtually all of the incoming sediment, resulting in a sediment deficit, loss of spawning gravels and degradation of the downstream channel.

A recent analysis of the stream-flow gauging stations downstream of 24 federal reservoirs in Kansas showed that most of the rivers had experienced substantial channel-bed lowering after installation of the dam (Juracek 2001). Another study of overflow dams in Kansas reported siltation of the channel bed upstream and increased channel bed and bank erosion immediately downstream of the structures (Juracek 1999). This resulted in the formation of depositional bars which diverted the flow toward or away from the banks causing bank erosion and channel widening. In a study of Bagnell Dam, in central Missouri, the tributaries located immediately downstream responded to lowered local base level by incising vertically, widening, and expanding headward (Germanoski and Ritter 1988). Changes continue as channel incision lowers water tables damaging the riparian area around the river, and reduces the number and duration of over bank flows (Power et al. 1996; Hart et al. 2002).

Other effects of a dam vary at different temporal scales and range from a narrowing or widening of the channel, and degradation or aggradation of the bed (Richard 2001). These opposite short-term responses in channel form appear to be related to discharge differences, reservoir surface area and type, and time-to-peak of the reservoir inflow hydrograph. Channel narrowing has been greatest below reservoirs that are large enough to contain floods (Kondolf 1997), but downstream channel changes are often confined to short reaches. In addition, impoundments with catchment areas that form less than 35% of the total drainage area are unlikely to induce an adjustment of channel morphology (Petts 1980b).

In other cases, the long-term response will likely be a complete alteration of channel morphology with a reduction of the cross-sectional area (Petts 1980c; Nichols et al. 2006). A reservoir in Wyoming reduced a multiple braided channel to a single channel (Patton and Hubert 1993). Reservoirs that reduce the frequency and duration of high flows have been reported to reduce lateral migration rates by factors of 3 to 6 (Shields et al. 2000). Other downstream changes that may occur include the loss of riffle-pool sequences, degraded and coarsened gravel size, armoring of the bed, bank collapse, and a meandering planform (Kondolf 1997; Hadley and Emmett 1998; Bednarek 2001; Assani and Petit 2004). The river becomes detached from its floodplain and its upstream reaches, with reduced recruitment of native species and encroachment of nonnative species (Poff and Hart 2002).

Channelization of most of the major rivers and their largest tributaries in western Tennessee during the 1950s and 1960s have caused stream bed incision that resulted in stream bank collapse and the introduction of large quantities of sediment (Shankman and Smith 2004). In Kansas, Juracek (2004) reported in detail on the effects of channelization on Soldier Creek. The effects varied within different portions of the river but generally included channel degradation that moved upstream resulting in bed elevation declines and bank failures.

Energy sources and relations

The construction of a dam on a river drastically changes the energy dynamics of the system. The primary carbon sources within the system change from allochthonous to autochthonous sources. Benthic algae decrease while phytoplankton types increase (Baxter 1977). A portion of this planktonic organic matter input sinks to the bottom of the reservoir, where it may escape oxidation and accumulate in lake sediments. As reservoirs fill in, these sediments act as a sink for carbon that has been removed from the atmosphere and fixed by photosynthesis on land and in the reservoir (Mulholland and Elwood 1982). Retention of these nutrients behind dams as a result of reduced velocity and longer residence time of water in the reservoir changes the availability of nutrients and composition of plant and microbial communities (Petts 1984; Gregory et al. 2002). Downstream areas suffer not just from this withholding of nutrients by the reservoir, but also from the elimination of bankfull flows that connect them with their riparian zone and floodplain (Magilligan et al. 2003). An additional impact of this disconnection may occur as shifts in the composition of streamside vegetation due to water stress (Smith et al. 1991). These same effects may occur as deepening and straightening of navigation channels reduces the ability of the river to overflow its banks and connect with its floodplain and riparian corridor.

Water quality

The predominant natural controls on the quality of river water are the climatic and geologic characteristics of the drainage basin. Water that is stored within a reservoir undergoes physical, chemical, and biological changes unlike those for rivers. Of particular importance in these changes are the forces that cause stratification, such as the loss of turbulent flow. Thermal stratification results in water discharge of different quality from different release elevations (Baxter 1977, Petts 1984). In thermally stratified reservoirs phytoplankton often proliferate in the warm epilimnion maintaining high concentrations of dissolved oxygen. In contrast, within the hypolimnion oxygen depletion may occur due to the settling of dead phytoplankton and the presence of heterotrophic bacteria. Consequently organic matter processing becomes anaerobic resulting in the production of hydrogen sulfide gas, release of carbon dioxide, decreased pH, and an increase in conductivity, alkalinity, and orthophosphate. Depending on the method of water release, the temperature of the water that flows downstream may be either colder or warmer than prior to reservoir construction (Baxter 1977; Bednarek 2001; Gregory et al. 2002).

A study of a major hydroelectric dam built on the White River in Arkansas revealed that the cold water release turned a pre-impoundment fish community from all warmwater species to one composed almost entirely of coldwater species including a loss of 77% of the fluvial specialists (Quinn and Kwak 2003). In addition, the decomposition of dense, submerged vegetation may create a high oxygen demand and high nutrient output in the discharged water. Hypolimnetic releases may have adverse effects on fish downstream of the dam either through dissolved oxygen deficits (Petts 1984) or the release of water containing supersaturated gases (Crunkilton et al. 1980).

Dredging associated with navigation channel maintenance may change water chemistry and circulation patterns at the site (Morton 1977). In addition, the water quality may be impacted by the release of sediments that are contaminated with heavy metals, chlorinated hydrocarbons and polynuclear aromatic hydrocarbons (Heath et al. 1999). Both commercial and recreational boating may also increase the amount and duration of suspended sediment particles and lead to sedimentation of backwaters (Smart et al. 1985; Waters 1995). Increased suspended sediment has been shown to cause decreases in the macrophyte community leading to reduced protection of the river bank from erosion (Garrad and Hey 1988).

Biotic interactions

The negative impacts of a dam on the biota of a river are well documented. Factors such as the alteration of the natural hydrologic and geomorphic regimes (Dynesius and Nilsson 1994; Moyle et al. 2003; Dieterman and Galat 2004; Quist et al. 2004); disruption of riparian plant communities (Dynesius and Nilsson 1994; Ward and Stanford 1995; Merritt and Wohl 2006); blocking of migration routes via physical and thermal barriers and loss of navigational cues (Dynesius and Nilsson 1994; Luttrell et al. 1999; Duncan 2002; Cooke and Leach 2004; Sanches et al. 2006; Schmetterling and McFee 2006); habitat fragmentation, loss of refugia and the associated isolation of populations (Dynesius and Nilsson 1994; Duncan 2002); loss of native fish populations due to either intentional or unintentional introduction of sport fish or other alien species (Winston et al. 1991; Phillips and Johnston 2004; Havel et al. 2005; Quist et al. 2005; Falke and Gido 2006); the mortality of larvae and juveniles at water intakes (Ferguson et al. 2006); and changes in the food resources all result in a decline in biodiversity and the alteration of natural food webs (Bednarek 2001; Poff and Hart 2002; Chester and Norris 2006). A study of biodiversity losses in the United States concluded that dams, impoundments, and other barriers impacted 17% of threatened and endangered species examined. This percentage increased to 30% of the examined species when agricultural diversions were included (Wilcove et al. 1998).

Studies have suggested specific changes in river biota that may occur as a result of damming. Interactions between inflow patterns, reservoir chemistry, and discharge regime were shown to have an important effect on the composition and biomass of filamentous and epiphytic algae below the Glen Canyon Dam in Arizona (Benenati et al. 2000). Within reservoirs, benthic invertebrates change to lentic types with an increase of mosquito breeding and other standing water creatures, and an increase in chironomids due to their ability to endure low oxygen concentrations that are likely to occur in new impoundments. Invertebrate diversity, richness, and total density decrease significantly below the dam, while total biomass has been found to increase or decrease (Baxter 1977, De Jalon et al. 1994; Nichols et al. 2006). In general, planarians, mayflies, beetles, stoneflies and caddisflies disappear or decrease in abundance (De Jalon et al. 1994; Tiemann et al. 2002), and there is a general downstream reduction of all benthic invertebrates with temperature dependent life cycles (Spence and Hynes 1971; Lehmkuhl 1972). Mussels may be eliminated not only from the impoundment, but also from the upstream and downstream areas, since they are dependent on flowing water for feeding and waste elimination, and because the dam is a barrier to fish that carry glochidia (Bogan 1993; Oesch 1995; Parmalee and Polhemus 2004). Additional problems for mussels caused by dams include changes in suspended and bed material load, bed composition, channel form, channel position, and channel stability (Brim Box and Mossa 1999).

Effects of large impoundments on the fish community include changes from benthic to lentic types (Yount and Niemi 1990; Winston et al. 1991; Taylor et al. 2001; Galat et al. 2005), invasion of tributaries by introduced fish species (Ruhr 1956) with increased predation of native species (Winston et al. 1991), loss of native species and genetic isolation due to disconnectivity (Winston et al. 1991; Luttrell et al. 1999; Wilde and Ostrand 1999; Holcik 2001; Tiemann et al. 2002, Galat et al. 2005), higher fish yields in new impoundments due to increased cover for young of the year by the flooded vegetation, increases in fish parasites, the elimination of migratory fish species (Baxter 1977), and the downstream elimination of coldwater fishes or warmwater fishes depending on the source of the dam's discharge waters (Spence and Hynes 1971). Upstream relative abundance and species composition have been shown to change dramatically after reservoir construction (Erman 1973) and the bioenergetics of all biota may be altered due to changes in water temperatures (Poff and Hart 2002).

Changes associated with impoundments are causing fish assemblages in North America to become homogenized, largely due to the introduction of macrohabitat generalists that prey upon or out compete the native fluvial species (Herbert and Gelwick 2003). Predation has been reported to be a significant factor in the reduction or loss of native fluvial species (Zale et al. 1989; Liechti 1994). A study in Indiana of tributaries upstream of impoundments showed higher numbers of piscivorous species and lower richness of fluvial species in the impounded streams versus the unimpounded (Guenther and Spacie 2006). In addition to the effects of direct predation, studies have shown that small cyprinids alter their behavior in response to predation by moving into shallower waters and decreasing foraging (Power et al. 1985; Mammoliti 2002).

A recent literature review of the potential impacts of small watershed impoundments on fish communities reported that "impoundments can reduce the quantity and quality of stream habitat, alter reproductive and feeding behavior of fishes, and increase the number and sizes of predatory fish and trophic generalists within a stream system" (Mammoliti 2002). Studies of fish assemblages in impounded and unimpounded headwaters of a river in Texas revealed that the lentic habitats of impounded lower reaches appeared to reduce movement by fluvial specialists among streams and, thus, reduced their opportunity to recolonize dry reaches. In addition, macrohabitat generalist species, which were abundant in the littoral zone of the impoundment, were able to recolonize or tolerate environmental conditions in these intermittent reaches (Herbert and Gelwick 2003). When impoundments block the movement of fluvial species from upstream areas, they prevent their use as physical and thermal refugia. Many fish are attracted to these groundwater-influenced areas for spawning. Blocking access may prevent spring spawners from laying their eggs in headwaters during high water, where they develop quickly as water levels recede and temperatures rise (Power et al. 1999, Meyer and Wallace 2001). Impoundments also interfere with the ability of fluvial species to utilize downstream reaches as refugia from drought and for feeding (Lienesch et al. 2000). When fluvial species move downstream into an impoundment they are put at increased risk when they are forced to deal with introduced predators and unsuitable habitat conditions (Winston et al. 1991, Herbert and Gelwick 2003). In addition, the blocking of migration routes can result in fragmentation of habitat with associated isolation of populations, preventing genetic exchanges between populations. This loss, along with the loss of critical habitat may increase the risk of extinction (Power et al. 1999, Meyer et al. 2003).

Both commercial and recreational navigation may impact fish communities through entrainment of larval and juvenile fishes by propellers (Bartell and Campbell 2000). Additional mortality may be related to stranding that occurs when the wake of large vessels cause a temporary drawdown (Adams et al. 1999). A study of the effects of recreational boating traffic on nest defense by longear sunfish showed that slow-moving boat traffic caused the males to leave their nests undefended, while fast-moving boat traffic caused increased turbidity creating more opportunity for predation of the nest (Mueller 1980).

Oil and Gas Production

Oil and gas production may degrade the water quality of impacted areas through several mechanisms and by-products. Substances that may be present in the gas, oil, wastewater or waste rock include toxic organic compounds such as benzene and naphthalene; dissolved solids such as sodium, bicarbonate, and sulfate; and metals or radionuclides (Otton et al. 1997). Surface and ground waters may be polluted either due to surface runoff, accidental spills, injection, waste water disposal, or leaching of contaminants from waste rock. Although accidental spills have received the most attention (Confluence Consulting 2004), water co-produced with oil and gas constitutes the single largest environmental impact for the oil and gas industry (Sirivedhin and Dallbauman 2004).

Other impacts from oil and gas production may result from the associated infrastructure development that can include roads and other transportation structures, well sites, utility lines, pipelines, and holding facilities for waste products (Confluence Consulting 2004).

Flow regime

Oil production has its greatest impact on the flow regime of streams through its interactions with groundwater supplies. Oil shale production uses large quantities of water during processing (Confluence Consulting 2004), and production of coalbed methane may extract large quantities of groundwater as a by-product (Griffiths and Woynillowicz 2003). A reduction in groundwater supplies shortens the length of the base flow for stream habitat maintenance, and reduces the dissolved oxygen and dissolved inorganic and organic nutrients necessary for stream productivity (Power et al. 1999). A recent study in Alberta, Canada, of the effects of conventional and non-conventional oil and gas production found that roughly 4% of all freshwater allocations were being used for oil extraction and processing. Of particular concern is the fact that a portion of this water is disposed of in deep geological formations where it is permanently removed from the hydrologic cycle (Griffiths and Woynillowicz 2003).
Flow regime may also be impacted directly when waste water from processes such as coalbed methane production are released directly into streams resulting in higher flows (Griffiths and Woynillowicz 2003). Increased flows may result in changes to the stream channel including channel and bank erosion, and substrate size; and changes in the biota.

Physical habitat quality

Discharge of waste waters into receiving waters from coalbed methane or other types of energy production may result in increased turbidity, sedimentation, and erosion leading to a loss of in-stream habitat (Griffiths and Woynillowicz 2003). Other associated activities such as infrastructure development of roads and buildings may have similar impacts due to landscape disturbance.

Energy sources and relations

As mentioned above, reductions in groundwater levels may impact stream energy sources by lowering dissolved inorganic and organic nutrients necessary for stream productivity (Power et al. 1999). Specific effects of various oil products on stream energy sources have also been reported. Laboratory studies have shown that with increasing concentrations of oil, algal cell counts and growth rates decrease (Mahaney 1994; El-Dib et al. 2001). Diesel fuel and lubricating oils, as well as the aromatic hydrocarbons benzene, toluene, and ethyl-benzene have also been found to cause reduced growth of freshwater algae (Dennington et al. 1975; Herman et al. 1990). Finally a study of oil shale water discharges in Estonia reported that the abundance of phytoplankton was negatively related to these discharges (Ratsep et al. 2002).

Water quality

Both conventional and non-conventional oil and gas production may affect water quality of the surrounding area. While underground pressure is usually adequate to ensure the flow of conventional oil and gas to the surface, the introduction of drilling fluid is required for some non-conventional methods. Drilling fluid is pumped in and then out of the well. This fluid not only contains the original additives, it may also be contaminated by underground chemicals it contacts such as heavy metals and hydrogen sulfide, making disposal problematic due to potential leaching to surface or ground waters (Confluence Consulting 2004).

Another by-product of these activities is highly saline water. In addition to using large quantities of water, conventional oil wells may produce large quantities of highly saline water or brine that is separated from the oil and pumped back underground into the oil-producing formation, or disposed of on the surface creating salt scars. A non-conventional source of oil—oil shale—also produces large volumes of saline water with elevated levels of several major ions and heavy metals. In addition to the problem of disposing of this water, the leaching into surface waters of salts, heavy metals, and organic compounds is a problem (Otton et al. 1997; Confluence Consulting 2004). Large quantities of fresh or saline water with high levels of total dissolved solids may be produced due to coalbed methane production. Disposal methods of this water vary depending on its salinity, but may include surface discharge to rivers, use of the water for crops or livestock, injection into groundwater, discharge to evaporation ponds, or deep well disposal (Griffiths and Woynillowicz 2003).

A study in northeastern Oklahoma looked at the effects of salt scars on associated streams. Surface waters exceeded total dissolved solid limits for drinking water. In addition these waters contained varying amounts of naturally occurring radioactive materials and trace elements such as arsenic, barium, selenium, cadmium, chromium, copper, lead, nickel, silver, zinc, mercury, lithium, and boron. This waste water may eventually find its way to groundwater supplies (Otton et al. 1997).

Increased fine sediment inputs to associated streams may occur from construction and operation of both oil and gas pipelines for transport of oil and gas. These inputs can result from trenching to lay pipeline beneath the stream channel, runoff from construction areas, erosion resulting from construction of related structures and stream bank modifications, erosion due to removal of the riparian canopy, or hydrostatic testing (Penkal and Phillips 1984). Impacts of fine sediment to streams may be reviewed in other portions of this document.

Biotic interactions

Oil interferes with the vital functions of many aquatic organisms. A coating of oil on plants may inhibit both respiration and photosynthesis, while for aquatic animals it may cover epithelial surfaces interfering with respiration ((Penkal and Phillips 1984). Substances in oil or its by-products have also been shown to impact aquatic communities. Exposure to these substances may result from pipeline breaks, transport accidents, leaching of stored materials, or through the disposal of waste water. Transport accidents are not uncommon. A study of incidents in Newark Bay and its major tributaries found that more than 1453 accidental spills had occurred between 1982 and 1991 with the bulk of the materials released consisting of petroleum products (Gunster et al. 1993).

Recovery by the invertebrate community of a river in Idaho took roughly 16 months after an unleaded gasoline spill (Pontasch and Brusven 1988). Recovery of the invertebrate riffle communities from two crude oil spills in Ozark streams due to pipeline breaks each took about one year (Crunkilton and Duchrow 1990; Poulton et al. 1998), while the invertebrate communities of backwater areas were still impacted after 18 months (Poulton et al. 1997). In Wisconsin, a train accident that released aromatic hydrocarbons into a river caused a major fish kill and tissue damage to the surviving fishes (Caldwell 1997).

In controlled studies, cutthroat trout exposed to various concentrations of Wyoming crude oil in water had slower growth, and gill and eye lesions, at discharge concentrations allowed within several western states (Woodward et al. 1981). Other studies found that exposure to No. 2 fuel oil causes decreased survival of juvenile rainbow trout (Steadman et al. 1991); while shale oils reduced the swimming capacity of squawfish and the richness of impacted invertebrate communities (Woodward et al. 1987). Leachate from these shale oils caused reduced growth of fathead minnows and reduced survival of mayflies (Woodward et al. 1985).

Exposure to both naphthalene and benzene in laboratory studies has been shown to decrease egg hatch and fry length, as well as mortality in rainbow trout and fathead minnows (DeGraeve et al. 1982); and waste water from oil refineries in Texas that contained these substance reduced fish diversity and abundance in a receiving stream (Kuehn et al. 1995). Fitness and growth of coho salmon fry decreased linearly with increased concentrations of toluene and naphthalene (Moles et al. 1981), while both juvenile and adult salmon have been shown to avoid aromatic hydrocarbons (Maynard and Weber 1981; Weber et al. 1981). Sediment contaminated with coal tar produced over 95% embryo and larvae mortality of shortnose sturgeon in Connecticut (Kocan et al. 1996), and effluent from oil production tainted with heavy metals caused stress in channel catfish (Martin and Black 1996). Studies of chromium exposure to a wide range of sensitive invertebrates resulted in reduced growth, inhibited reproduction, and increased bioaccumulation; as well as reduced growth of rainbow trout and chinook salmon fingerlings (Eisler 1986).
A study of streams in Oklahoma that received oil refinery effluent or oil field brine reported that the effluent-impacted stream had lower invertebrate richness and diversity (Mathis and Dorris 1968), but a related study reported that with exposure to higher concentrations of oil field brine, invertebrate diversity decreased due to the loss of sensitive species such as mayflies (Harrel and Dorris 1968). In a study of the impacts of oil field brine in streams of eastern Kentucky, researchers found spatial changes in structural and functional characteristics of both the fish and

macroinvertebrate communities. However, the authors suggested that since the fish seemed to be more tolerant of salt, changes in the fish community might be in response to changes in the invertebrate community (Short et al. 1991).

Thresholds

No threshold levels for oil or gas impacts were found in the literature. Suter (1997) noted that "In general, petroleum and its constituents have low toxicity relative to other classes of chemicals that raise environmental concerns such as pesticides, heavy metals, and chlorinated diaromatic hydrocarbons. In addition, most petroleum constituents are relatively nonpersistent, and petroleum hydrocarbons do not tend to biomagnify through food webs." Another review suggested that non-conventional energy production tends to have greater environmental impacts than production of conventional resources (Confluence Consulting 2004).

Military Installations

Military installations provide a unique situation for assessing the effects of land disturbance on streams. Most contain large tracts of land that are free of urban or agricultural lands due to their use as military training areas. Streams within these areas may be exposed to a wide range of activities including light infantry maneuvers, bombing practice, and mechanized forces practicing river crossings. Depending on the size of the operation, loss of vegetation, soil compaction, and sediment runoff may occur at the local or watershed scale as in urban and agricultural uses. However, because these areas suffer repeated disturbances and remain denuded for years through repeated training, the effects of these land disturbances may be more pervasive than those associated with urban and agricultural land use (Quist et al. 2003; Maloney et al. 2005). At Fort Benning in Georgia, the main disturbance to streams was fine sediment that resulted from military training activities and unpaved roads within the watershed (Maloney and Feminella 2006).

Flow regime

Increased soil compaction either from military training or base facilities and roads should have impacts similar to other disturbances discussed such as increased runoff resulting in higher peak discharge, flood frequency, and flow variability (Poff et al. 2006).

Physical habitat

Studies of the effects of military training activities at Fort Riley on impacted streams in northeastern Kansas showed increased sediment in stream pools and riffles (Quist et al. 2003). At Fort Benning, researchers compared the effects of natural geomorphic and topographic variables versus military training disturbances and found that training disturbances had stronger relations to physical habitat variables. Streambed instability was related to the percentage of the watershed containing disturbed land, while quantities of coarse woody debris, benthic particulate organic matter (BPOM), and dissolved organic carbon (DOC) were negatively related to the percentages of bare ground and road cover in the watershed (Maloney et al. 2005).

Energy sources

Another study at Fort Benning looked at the effects of upland soil and vegetation disturbance on rates of stream metabolism (Houser et al. 2005). Ecosystem respiration was negatively related to the amount of watershed disturbance in three of four seasons. It was also positively related to the amount of coarse woody debris, which along with BPOM and DOC was negatively related to percentages of bare ground and road cover. Gross primary production was low in all treatments and not related to disturbance levels.

Water quality

In a related study at Fort Benning, researchers looked at the effects of upland soil and vegetation disturbance on stream chemistry (Houser et al. 2006). They found that during baseflow the mean total and mean inorganic suspended sediment concentrations increased significantly with increasing watershed disturbance, while dissolved organic carbon and soluble reactive phosphorus concentrations decreased. A study of water chemistry at Fort Benning found that total Kjeldahl nitrogen and total organic carbon were negatively related to the amount of military activity in the watershed (Bhat et al. 2006). They also reported significant positive relations between road density within the base and both total phosphorus and chloride.

A study at Fort Riley in Kansas looked at stream crossing materials and resultant turbidity (Sample et al. 1998). It revealed that turbidity levels, total solids, total dissolved solids and total suspended solids were all higher for earthen fords crossing streams than hardened fords.

Biotic interactions

Researchers quantified military disturbance based on the quantity of bare ground on slopes >3% and as unpaved road density to look at the effects of these disturbances on the macroinvertebrate community at Fort Benning (Maloney and Feminella 2006). Macroinvertebrate assemblages were associated with watershed disturbance; with decreases in EPT and chironomid richness, percentage clingers, and the Georgia stream condition index as disturbance levels increased. At Fort Riley, military training levels were related to the fish community. As training intensity increased, there was a concurrent reduction in abundance of benthic insectivores, herbivore-detritivores, and silt-intolerant species. Study sites with the highest training levels were characterized by high numbers of trophic generalists and other tolerant species (Quist et al. 2003).

Thresholds

Maloney and Feminella (2006) reported that the results of their work at Fort Benning suggested that a threshold for stream disturbance by military installations would be met when 8 – 10% of the watershed consisted of bare ground and unpaved road cover.

Mining

All types of mining may have negative effects on streams. Sand or gravel extraction may directly release large quantities of sediments into stream channels, while mining has the potential to produce large quantities of tailings and other waste that may eventually reach a stream through erosion or dam collapse (Waters 1995). Instream gravel mining typically occurs in the active channel of the river because substrates in these areas are usually durable and well-sorted (Kondolf 1994). The mechanical extraction in the active channel results in the release of fine sediment and the destabilization of the stream channel which may damage public infrastructure such as bridges and pipelines (Roell 1999). Mining of this type can result in increased turbidity, suspended sediment, and bedload (MacDonald et al. 1991). Changes in channel morphology can include a decrease in the channel cross-sectional area due to sedimentation, or an increase in cross-sectional area, slope, and velocity, resulting in headcutting of the channel (Kanehl and Lyons 1992; Jacobson et al. 2001). Channel incision can occur over great distances both upstream and downstream of the mine sites (Roell 1999).

Although mining can result in excessive sediment inputs, this problem can be avoided with proper management practices, leaving alteration of stream chemistry as the most serious threat. These changes in water chemistry depend on the type of mining and the extraction process, but in most instances, both pH and conductivity are affected (MacDonald et al. 1991). In some instances, toxic substances such as metals and sulphuric acid may be released into streams as precipitates or be adsorbed on sediment particles and released at a later time (Waters 1995). One way this can occur is through acid mine drainage (AMD) which occurs when water moves through mines and mine tailings prior to reaching a stream. There it mobilizes toxic metals and combines with sulfur-bearing minerals to form solutions of sulfuric acid. Resultant impacts to aquatic ecosystems from AMD may be severe due to acidification, metal precipitation smothering stream substrates, and sediment or water toxicity in association with trace metals (Hoiland et al. 1994; Soucek et al. 2001).

Flow regime

Reports of the effects of mining on flow regime are limited. Declines in groundwater levels, diversion of surface drainage, and reduced flow variance due to surface coal mining, and altered quantity and variability of stream flow due to underground coal mining have been reported, along with increased flood risk due to channel instability from sediment extraction (Bevans et al. 1984; Borchers et al. 1991; Kondolf 1994; Temple 1997; Kondolf et al. 2002; Chaplin 2005).

Physical habitat

Streams from which sand and gravel are extracted may suffer severe impacts (Waters 1995). Studies in three Ozark streams found that gravel mining resulted in increased sediment loads and altered geomorphology (Brown et al. 1998). Affected reaches had increased bankfull widths, lengthened pools, and decreased riffles. Research done in California found that gravel mining changed local channel morphology by means of bed and bank erosion, channel incision, and coarsening of bed material. It also accelerated distant channel bed and bank erosion, and undermined bridges and other structures through alteration of sediment transport leading to reduced sediment delivery (Kondolf 1994; Kondolf 1997). Other changes in channel morphology may include increases in slope and cross-sectional area of affected streams due to direct excavation which may result in decreases in embeddedness, cover and large woody debris retention (Jacobson et al. 2001). In Europe, sediment mining of alluvial rivers has been reported to produce later channel instability, bed armoring and channel incision which alters the frequency of floodplain inundation (Rinaldi et al. 2005).

Reports of impacts due to rock or mineral extraction include stream channel subsidence in Utah from longwall mining of coal. Subsidence resulted in short-term channel changes that led to increases in pool length, numbers and volumes, an increase in median particle size of pool substrate, and some constriction in channel geometry (Sidle et al. 2000). Studies of coal mining impacts to tributaries in Virginia watersheds reported sedimentation as the major impact, along with increased embeddedness in shallow-water habitats and increased fine sediment depth in pools (Temple 1997).

Energy sources

The majority of studies looking at the impacts of mining on energy sources involve the effects of mine drainage. In West Virginia, periphyton biomass and leaf decomposition was significantly reduced in streams affected by AMD (Simmons et al. 2005), as was species richness and diversity of periphyton and macroalgal communities in Ohio (Verb and Vis 2001; Verb and Vis 2005). Studies in Colorado mountain streams revealed that the breakdown of plant litter decreased with increasing products of mine drainage including increased concentrations of zinc and increased deposition rates of metal oxides. Fungal and algal diversity also decreased, although fungal biomass was stable, while microbial respiration was negatively related to the deposition rates of metal oxide (Niyogi et al. 2001; Niyogi et al. 2002a; Niyogi et al. 2002b). Others have reported that large influxes of dissolved metals resulted in decreased chlorophyll a content of periphyton (Hill et al. 2000).

A study of gravel mining in Ozark streams found that the transport of fine particulate organic matter from riffles to pools decreased (Brown et al. 1998), while a study of mining-induced sedimentation in Alaska reported a reduction in both biomass and nutritional value of stream periphyton (Van Nieuwenhuyse and LaPerriere 1986).

Water quality

Effects of mining-related extraction activities primarily come from two sources: increased sediment discharges into streams from present-day operations (Duchrow et al. 1980; Duchrow 1982; Duchrow 1983); and runoff waters leaching metals and chemicals into streams from abandoned mines and ore tailings (USFS 1999). Pollutants such as cadmium, zinc, sulfate, and sediment from mining and the wastes associated with extraction, have been reported as having impacts on receiving streams in both Kansas and Missouri (MDNR 1998; USEPA 2004; Anonymous 2005).

Brown and others (1998) reported that gravel mining in Ozark Plateau streams significantly altered the turbidity during, but not after, gravel mining. Turbidities in areas affected by gravel mining doubled during extraction. Studies of the effects of lead mine tailings released into nearby Ozark streams showed significantly higher turbidities two miles downstream three days after a dam collapse (Duchrow et al. 1980). Another study of dam failures for two barite tailing ponds in the same region found significantly greater turbidities for either four or 14 days after the event, and during high flow periods for several months (Duchrow 1982). Sediment from dam failures of barite tailing ponds has been reported to cause fish kills (Duchrow 1982; Smith 1988). Data from the early 1980's on strip-mining of coal near Keoto, Missouri showed that SS samples taken during one 26-hour storm flow showed a receiving stream had mean suspended sediment concentrations of 13,500 mg/L and a maximum concentration of 19,900 mg/L. A similar stream nearby on the same dates had a mean suspended sediment concentration of 2,000 mg/L (Blevins 1986). These values far exceed those reported to cause impairment to fish communities (Doisy and Rabeni 2004).

Changes in water chemistry from runoff waters or leachate may impact receiving streams and groundwater supplies. Acidification and elevated sulfate are two of the most common impacts (Gray 1996), but in West Virginia concentrations of Ca, Mg, K, and Na were reported to be 2 – 4 times higher in associated surface waters along with increased mean specific conductance and mean dissolved solids concentration (Borchers et al. 1991). A study in Indiana of AMD found elevated SO4, Fe3+, Al Fe2+, Ca, Mg, Na, Cl, Mn, K, Si, Zn, Ni, Pb, Cd, V, Be, and Cr, with several of these contaminants exceeding local and/or national surface water quality standards (Brake et al. 2001). In Missouri, lead and zinc mining have the most significant effects on water quality (Smith 1988), with reports of abandoned mines in the southeastern part of the state causing elevated levels of Pb, Zn, Ca, and Mg in the water and the aquatic biota (Jennett et al. 1981).

Biotic interactions

Many studies have linked mining with the degradation of invertebrate or fish communities in associated streams (Matter and Ney 1981; Griffith et al. 2001; Courtney and Clements 2002; Diamond et al. 2002; Fernandez-Alaez et al. 2002; Bruns 2005; Locke et al. 2006; Tripole et al. 2006). However, assessing the specific effects of mining on biological communities may be complicated by multiple impacts such as increases in turbidity, sedimentation, acidification and heavy metals accumulation (Ward 1984; Hoiland and Rabe 1992). The effects of increased turbidity on the biotic communities of streams have been extensively documented, and include reduced primary production, changes in plant and invertebrate community composition and abundance, and behavioral, sublethal, and lethal effects in fishes (Doisy and Rabeni 2004). In Missouri, severe damage to invertebrate and fish communities due specifically to mining-related turbidity has been reported as a result of tailing pond collapses (Duchrow 1982; Duchrow 1983). Mining-related sedimentation has been reported to decrease densities of silt-sensitive fish species in Ozark streams degraded by gravel mining (Brown et al. 1998). In Virginia, a study of the impacts of coal mining found that taxonomic metrics of the fish community were not significantly related to quantities of sediment in shallow-water or pool habitats, but functional metrics were related (Temple 1997). With increasing sedimentation, relative abundance of omnivores increased while insectivore and top carnivore abundances decreased.

The effects of acidification and metal toxicity on the aquatic biota of mining-impacted streams have been heavily researched. Total invertebrate richness, genetic diversity, mayfly density and richness, collector-gatherer richness, and scraper density and richness were positively correlated to stream pH in studies of streams in eastern states (Smith et al. 1990; Rosemond et al. 1992). Constant acid mine drainage was found to eliminate the mayfly and stonefly fauna in western Pennsylvania streams (Roback and Richardson 1969), and high densities of chironomids have been reported (Gray 1996). Studies in acid streams of Ontario showed that communities were greatly simplified with reductions in mayfly and stonefly richness (Mackay and Kersey 1985). A comparison of reference streams to treated and untreated AMD streams found that untreated AMD streams had significantly lower invertebrate density and diversity than the reference streams, and even treated AMD streams had lower diversity (Simmons et al. 2005). A study of coal surface mining in Alabama compared the stream of an active mine to that of a reclaimed mine and one with no mining history. Leaf packs that were placed in the reference stream revealed an abundant and diverse insect community in contrast to the active mine stream which contained no shredder or collector species. Invertebrates in the reclaimed mine stream were intermediate in abundance and diversity to the other two streams (Scheiring 1993). Another concern is that recovery of the invertebrate community to acidity may be slow. A study in Montana found that macroinvertebrate communities were still in the recovery process more than five years after major improvements to the wastewater treatment at a mining facility on Silver Bow Creek (Chadwick et al. 1986).

Laboratory studies of the effects of acidic water on young-of-the-year smallmouth bass suggested that environmental acidity was sufficient to cause losses in this species, even in the absence of synergists such as heavy metals (Hill et al. 1988). However, other studies of the effects of acidification on adult stream fish in New York found that the total biomass of fish communities was not seriously affected at moderately to strongly acidified sites, while species richness and total density of fish were adversely affected at strongly to severely acidified sites (Baldigo and Lawrence 2000). Although streams in the Ozark Highlands are well-buffered, and alkaline relative to those in the northeast, Matthews (1998) noted that he typically found greater densities of fish in the more alkaline Ozark mountain streams, than in their less alkaline Ouachita counterparts (Matthews 1998).

Studies of the effects of AMD on fish communities have found the same effects. In Tennessee, streams impacted by abandoned coal mine drainage were characterized by low fish species richness and abundance. The acidified streams were dominated by centrarchids, while nearby reference streams were dominated by cypriniids (Schorr and Backer 2006). Tests of bluegill mortality in AMD mixing zones of Alabama streams showed that these areas of the stream may lower the survival of fishes, reduce available habitat, and impede movements of migratory fish (Henry et al. 1999).

Heavy metals in both the water and sediment may have a negative impact on the biotic communities. A study located downstream from a hard rock mine in Colorado, showed concentrations of As, Co, and Cu were significantly elevated in the water, sediments and invertebrate tissues. The invertebrate community was severely impacted with reductions in both biomass and species richness (Beltman et al. 1999). Other studies have also shown that metal toxicity affects many types of aquatic invertebrates with reductions in overall taxonomic richness and density. Mayflies and stoneflies have been found to be particularly sensitive (Quinn et al. 1992; Hoiland et al. 1994; Malmqvist and Hoffsten 1999; Richardson and Kiffney 2000; DeNicola and Stapleton 2002; Maret et al. 2003). Impacts to fishes by heavy metals from mining are also reported. Studies have shown decreases in enzyme and ALA-D activity (Schmitt et al. 1984; Dwyer et al. 1988); reduced testosterone levels, body weight, and survival (Dube et al. 2005; Dube et al. 2006); and reductions in species richness (Maret and MacCoy 2002).

Thresholds

There are few studies of threshold levels of mining, but some studies have reported comparisons between the impacts on streams of mining versus other land use activities. A study in Virginia using GIS to quantify land use in various tributaries reported that tributaries draining mining-influenced watersheds had lower benthic invertebrate scores than streams in agricultural or forested watersheds (Locke et al. 2006). In the eastern United States, a study investigating the impacts of land-use patterns on benthic invertebrates found that coal mining had a stronger negative impact than agriculture or low intensity rural-residential patterns (Chambers and Messinger 2001). Similar results were found when a variety of land use patterns including mining, forest, urban, and agriculture were investigated in northeastern Pennsylvania and New York (Bruns 2005).

Timber Harvesting

Timber harvest and related road development may alter a stream’s hydrology, sediment budget, nutrient cycles, and morphology (Bilby et al. 1989; Jones and Grant 1996; Christie and Fletcher 1999; Jacobson and Gran 1999; Beschta et al. 2000; Bowling et al. 2000; Swank et al. 2001). Changes in upland areas of a watershed usually have two effects: an increase in runoff and an increase in sediment yield; while the clearing and compaction of riparian areas in direct proximity to a stream include increased sediment yields, and decreased flow resistance and erosional resistance of the streambed and banks (Jacobson et al. 2001). These changes may affect the aquatic biota either indirectly through alteration of in-stream habitat or directly through the addition of sediment to the stream channel or alteration of flow.

Watershed and riparian removal of the vegetative cover, along with soil compaction and exposure may result in significant soil losses which enter associated streams (Scott and Udouj 1999). A study of various forestry practices in the southeast reported soil losses for up to four years after harvesting (Dissmeyer 1980). Significant sediment loss was also reported in Arkansas by following clearcutting with mechanical treatments (versus chemical) (Beasley and Granillo 1985). Recognizing the variable effects of timber management practices on different soil types, the Missouri Department of Conservation reported that various forms of cutting in Madison County, Missouri, resulted in different amounts of exposed soil. Three clearcuts, two pulpwood cuts and one conventional sawlog harvest were estimated to have produced 33%, 22% and 12% exposed soil, respectively (Barnickol et al. 2001).

Studies performed in the Ouachita Mountains looking at the effects of road building found that the amount of erosion was related to the amount of soil disturbance, the size of the area draining onto the roads, the soil erosiveness, the slope and slope length, and the amount of rainfall shortly after harvest (Scoles et al. 1994). Other studies from western, mountainous regions have indicated that the associated road building is more damaging than the actual cutting (Brown and Krygier 1971; Megahan and Kidd 1972; Schnackenberg and MacDonald 1998; Christie and Fletcher 1999). Sediment may erode from the road surface, road fills, and slope failures associated with road building long after regrowth of stabilizing vegetation (Megahan 1978; Reid 1984; Duncan et al. 1987; Megahan and Bohn 1989; Wemple et al. 1996; Jones et al. 2000; Wemple et al. 2001).

Flow regime

Changes in the flow regime of watersheds impacted by timber harvest are similar to those resulting during the construction phase of urban development. Timber harvesting both within the uplands and along the riparian corridors of streams may result in changes in the water yield, peak flows, water quality, and sediment yield, as a result of increased amounts of surface water flowing into a stream due to reduced evapotranspiration and increased runoff (MacDonald et al. 1991; Jacobson and Gran 1999; Jacobson et al. 2001; Swank et al. 2001).

Physical habitat

Changes occur in the channel morphology of streams impacted by timber harvesting due to the flashy nature of the hydrograph, increased erosion and deposition, and the loss of the stabilizing effect of vegetation (Smith et al. 1993; Ralph et al. 1994; Fetherston et al. 1995; McKenney et al. 1995; Birkeland 1996; Simon et al. 1999; Jacobson et al. 2001). An increased sediment load is usually the most damaging effect of forest management activities (MacDonald et al. 1991). Large increases in sediment entering the stream channel can result in the loss of animal habitat by increasing riffle area while filling in and reducing pool area (Lisle 1982; Ralph et al. 1994; Waters 1995; Woodsmith and Buffington 1996). This alteration of in-stream habitat may also result in a reduction of residual pool heterogeneity, an in-stream variable closely linked with stream habitat degradation (Waters 1995). This was confirmed in a study of headwater streams impacted by timber harvest in the Ozark Highlands. Increased timber harvest was significantly related to decreased residual pool heterogeneity and to an increase in mean riffle length (Doisy et al. 2000). Sediment increases from timber harvest or associated road construction may also cause changes in channel shape, sinuosity, and bed material size (Roberts 1987; MacDonald et al. 1991; Dose and Roper 1994), while effects of unbuffered stream bank include increased sediment yield and a decrease in channel cross-section and bankfull depth and width (Doisy et al. 2000; Jacobson et al. 2001).

Energy sources and relations

Timber harvest has a major impact on the nutrient cycle of associated streams due to increased water temperatures, increased primary production, and the long-term loss of leaf litter and woody debris, which may act as a food source or substrate for invertebrates while promoting benthic particulate organic matter retention (Bilby 1981; Webster and Waide 1982; Silsbee and Larson 1983; Lynch et al. 1985; Webster and Benfield 1986; Golladay et al. 1987; Webster et al. 1990; Binkley and Brown 1993; Wallace et al. 1993; Davies and Nelson 1994; Webster et al. 1994;

Webster and Meyer 1997; Bunn et al. 1999; Mitchell 1999; Steel et al. 2000; Kedzierski and Smock 2001).

Water quality

Forestry practices alter the chemistry of associated streams. Undesirable changes in stream temperature and concentrations of dissolved oxygen, nitrate-N, and suspended sediments may occur. Water temperature may increase by 3 – 7 °C with removal of the riparian canopy leading to increases in primary production and decreases in dissolved oxygen levels. Addition of large quantities of fine organic matter may add to the depletion, while sedimentation of the streambed may reduce oxygen diffusion into spawning beds. Concentrations of phosphate and nitrate-N usually increase after forest harvesting and fertilization adding to increased primary production and altered food webs (Binkley and Brown 1993). Increased fine sediment is likely to occur with mechanical forestry practices such as skidder activity, but general agreement exists that the highest erosion rates are due to the construction and use of roads to access the harvest (Kreutzweiser and Capell 2001; Swank et al. 2001). Although sediment increases from road building tend to lessen shortly after completion, cumulative increases in sediment yield may occur downstream for many years due to the lag between pulsed stream inputs and sediment routing (Swank et al. 2001).

Biotic interactions

Increased sediment load not only reduces preferred habitat for the biota (Waters 1995), at the local level fine sediments may result in the loss of preferred invertebrate habitat by increasing substrate embeddedness and altering the substrate particle size distribution (Minshall 1984). A summary of the effects of sediment upon benthic invertebrates in Piedmont streams of North Carolina (Lenat et al. 1981) indicated that with the addition of small amounts of fine sediment, densities of sediment-intolerant orders (EPT) might decrease without a resultant change in diversity, due to the reduction of the interstitial habitat. Additional increases that resulted in a more drastic change in substrate type could cause changes in the numbers and type of taxa (less EPT taxa and more chironomids). Other studies looking at the effects of fine sediment on community metrics in streams of the Ozark Highlands and the Appalachians reported inconsistent results (Angradi 1999; Zweig and Rabeni 2001).

The effects of logging on stream biota have been reported from regions of the United States and Canada. In Georgia, streams impacted by forestry were shown to have higher macroinvertebrate abundance, biomass and production with changes in the relative proportion of functional feeding groups (Stone and Wallace 1998), while impacted streams in Oregon had lower B-IBI scores (Fore et al. 1996). A study in Montana reported that a stream impacted by timber harvesting had lower aquatic biointegrity than did a similar one in a wilderness area (Rothrock et al. 1998). Juvenile chinook salmon in wilderness areas of Idaho and Oregon had higher mortality in areas of high road density (Paulsen and Fisher 2001). In Alberta, bull trout abundance was negatively related to levels of forest harvesting, while occurrence was negatively related to the cumulative area of the watershed harvested and its road density (Ripley et al. 2005). However, differences in stream size and physiography may render these results inapplicable to streams within the Midwest.

Generally, there is little published information on the effects of timber harvesting and/or associated roads on stream biota within the Midwest. One study in the Ouachita Mountains of Arkansas, reported that in small hydrologically variable streams, logging along with drainage basin differences had some influence on physical stream features, but that most of the differences in the fish and macroinvertebrate communities appeared to be related to basin differences or natural variability (Williams et al. 2002). In contrast, another study in the Ozark Plateau of Missouri looking at the effects of timber practices on aquatic macroinvertebrates of small streams reported that none of the invertebrate community metrics had any relation to watershed size. Instead, they were significantly related to local-level variables potentially affected by timber management such as substrate size (especially those < 2 mm in size) and organic matter; reach-level variables such as stream bank erosion and canopy cover; and watershed-level variables such as road densities and unbuffered stream bank (Doisy et al. 2000).

Transportation

Approximately 20% of land area in the United States is impacted ecologically by the presence of public roads (Forman 2000). Roads have wide ranging short- and long-term impacts on the biota, water quality, and physical habitat of stream ecosystems (Trombulak and Frissell 2000; Wheeler et al. 2005). Three stages of road construction act upon associated streams. These stages are the initial construction, presence, and the urbanization that follows. Effects of the actual construction are usually short-lived unless associated stream modifications are made such as channelization or stream bank stabilization. However, the usage of the road and the resulting urbanization that comes with access to new areas has long-term, chronic impacts (Angermeier et al. 2004). Here we address the first two stages. For the effects of the later stage see the section of this review entitled “Urbanization.”

Generally the most important impacts of roads on streams are the introduction of heavy metals and fine sediments along with alterations in the channel morphology, stream bank, and water quality (Little and Mayer 1993; Forman and Alexander 1998). During extreme storms, a road network may have major impacts on streams far from the actual event (Wemple et al. 2001). In addition, the construction of bridges or culverts that allow roads to cross streams may destabilize the channel leading to changes in bed morphology and blocking passage of the aquatic biota and woody debris (Gibson et al. 2005; Wheeler et al. 2005).

Flow regime

Water runoff is one of the key physical processes by which roads affect the stream and its biota. Effects vary widely depending on road size and proximity to the stream, watershed size, slope, soil type, riparian cover, and the size of the receiving system. Increased runoff may lead to reduced percolation of rainfall through the soil and subsequent reduction of groundwater levels, along with higher peak flows (Forman et al. 1998). Flooding frequency has been correlated with the percentage of road cover in a watershed (Jones and Grant 1996; La Marche and Lettenmaier 2001), but stream bank erosion is a more common result. In addition, the arrangement of the road network in relation to the stream network can influence interactions between peak flows and debris flows (Jones et al. 2000). No information on possible impacts to flow regime was found for airports or rail yards. However, these types of facilities are likely to have the same sorts of impacts as other large areas with impervious surface.

Physical habitat quality

Streams near major road construction are frequently channelized leading to increases in peak flows and channel slope, reductions in base flows, altered substrate composition, and reduced linkage to the floodplain. Higher peak flows, either from increased runoff or channelization, can alter channel morphology and the arrangement of woody debris, boulders, fine-sediment deposits, and pools. Changes in channel grade from channelization, bridges or culverts may cause local scouring, while the fixed location of these types of modifications prevents natural channel migration (Wheeler et al. 2005). Bridges and culverts fragment the channel, and the associated scour may result in higher width-depth ratios further downstream (Chin and Gregory 2001). Roads high on hill slopes may concentrate runoff forming new headwater stream channels. This leads to a more elongated network of 1st order streams that can impact erosion and sedimentation rates further downstream (Forman et al. 1998). Highway maintenance may also have negative impacts on the physical habitat of streams. In Colorado a study of traction sand introduced from nearby I-70 showed that the impacted stream had only one third the number of pools as two similar reference streams (Lorch 1998).

Energy sources and relations

Little information was found directly linking roads or road construction to changes in the energy sources of associated streams. However, changes in flow from increased runoff, increases in fine sediment due to nearby construction, and the removal of the riparian canopy at bridge and culvert construction sites would be expected to impact both primary and secondary production. One study in the UK reported that highway runoff in the form of total hydrocarbons, aromatic hydrocarbons, and heavy metals did not have a significant impact on epilithic algae but that the diversity of the aquatic hyphomycete (fungi) assemblage decreased at the most impacted sites. They also reported a reduction in leaf litter processing and a change in the invertebrate community from one dependent on benthic algae and CPOM to one dependent on FPOM (Maltby et al. 1995). Road salts have also been suggested as a possible cause of benthic algal mat formation in the Bow River, Montana (Schindler 2000).

Water quality

Impacts to water quality from roads and other transportation structures are generally from materials introduced into the system via runoff. The wide-ranging effects on the stream and its biota of fine sediment from road construction and related structures have been well documented for forest roads. Less information exists for larger roads. A study in the Chattooga River watershed of Georgia (Pruitt et al. 2001) showed that road density and associated sediment sources accounted for 51% of the total sediment load during storm events. Although most of these roads were located in heavily forested watersheds, there was no indication as to the type or use.

Highway construction along a stream in southern Ontario, resulted in an increase in suspended solids to more than 1000 mg/L as compared to normal levels of < 5 mg/L, and the unprotected banks continued to erode at an accelerated rate after the completion of construction (Barton 1977). In Florida, highway construction resulted in significant increases in both turbidity and suspended solids in the associated stream (Burton et al. 1976), and increases as high as 426% have been reported in a stream in Hawaii (Wong and Yeatts 2002), as has exceedance of state water-quality standards for suspended sediment (Hill 1996). In the Rocky Mountains, construction of a highway over a stream resulted in suspended solids below the construction site that were as much as 40 times natural levels, but levels returned to normal within 2 weeks after completion (Cline et al. 1982). In contrast sedimentation effects may be long-lived. A study in Ontario reported increases in bedload and sedimentation that lasted as long as nine years following highway construction (Bowlby et al. 1987). A study in California found that road density was positively related to embeddedness within the Russian River (Opperman et al. 2005). Application of road traction sand can also have a serious impact on associated streams. Sand applied to I-70 on Vail Pass in Colorado resulted in increased bedload and 150 times the background levels of sand in the impacted stream. In addition, the median substrate size of impacted riffles was 3.5 mm versus 38 mm for riffles in two reference streams (Lorch 1998). Installation of stream crossings, bridges and culverts have also been reported to increase fine sediment introduction (Taylor et al. 2004). Both suspended sediment and bedload increased after the construction of an unsealed stream crossing in Australia (Lane and Sheridan 2002).

In addition to increasing the fine sediment loading to streams, road construction, use, and maintenance also increases the amount of oil products, lead, salt, and other chemicals that make their way into associated streams. A major source of chloride to stream systems is deicing salt. Significant increases in both sodium and chloride have been reported from streams impacted by highways in Ontario (Bowlby et al. 1987), Colorado (Lorch 1998), Massachusetts (Lent et al. 1998; Mason et al. 1999), Pennsylvania (Koryak et al. 2001b), New Hampshire (Rosenberry et al. 1999), and New York (Godwin et al. 2003); and road salts have contaminated the groundwater in Toronto (Howard and Haynes 1993; Williams et al. 2000). Deicing agents from an airport in Pennsylvania were found to be the principal impact on a nearby stream (Koryak et al. 1998). The deicing reagents exerted a strong biochemical oxygen demand and led to elevated concentrations of ammonia. Deicing agents have also been reported to increase the mobility of heavy metals in the soil (Forman et al. 1998).

Other chemicals may be introduced into streams from road runoff and remain at elevated levels. Nickel, chromium, lead, and copper are sloughed from brake linings, while zinc, lead, chromium, copper and nickel may emanate from tires (Davis et al. 2001; Paul and Meyer 2001). Chemicals that have been reported to be at elevated levels due to proximity to highways include elevated Ca, K, Mg, PB, Zn, Cu, Cr, Cd (Van Hassel 1980; Forman et al. 1998; Mason et al. 1999; Callender and Rice 2000), and polycyclic aromatic hydrocarbons (Shinya et al. 2000). Another possible source of chemicals in streams may emanate from the transportation of hazardous materials. While only a small fraction of the materials transported are spilled, these incidences are more likely to occur on bridges (Mattson and Angermeier 2007).

Biotic interactions

Several studies have documented significant relations between the density of forest roads and various aquatic communities (Reid 1998; Baxter et al. 1999; Paulsen and Fisher 2001; Doisy et al. 2002; Hughes et al. 2004; Ripley et al. 2005). In contrast, few studies have documented relations between highway density and stream biota. One study in Arkansas reported that overall ecological integrity of streams declined as road densities increased (Radwell and Kwak 2005). More specific studies in that state reported that both road density and percentage of urban land use had negative impacts on various components of the IBI (Dauwalter et al. 2003), and the reconstruction of a state highway caused a shift in the feeding regime of fishes in an impacted stream (Ebert and Filipek 1988). Decreased EPT richness and increased densities of some chironomids were found in eastern streams with an increasing gradient of watershed road density and percentage of transportation land use (Kratzer et al. 2006), while in Montana, a study of land use variables showed that road density had the highest number of significant correlations with fish IBI metrics (Bramblett et al. 2005).

As with any construction near a stream or river, the introduction of fine sediment due to highway construction has a major impact on the aquatic communities. A study of bridge construction in Panama suggested that the invertebrate community structure changed due to the associated erosion of the bottom sediments (Blettler and Marchese 2005), and reduced diversity and abundance of invertebrates have been reported due to sedimentation from road and bridge construction in Nigeria (Ogbeibu and Victor 1989). Fine sediment introduction from road construction in Ontario resulted in changes in the invertebrate and fish communities that lasted as long as five years after construction was completed (Taylor and Roff 1986). Bowlby and others (1987) also reported long-term impacts (nine years after completion) related to the construction of a highway in Ontario. Modifications to the stream channel resulted in elevated bedload, lower invertebrate density due to sedimentation, and lower biomass of brook trout.

Other modifications due to highway construction, such as channelization to a stream, have been found to lower density and biomass of the fish community (Oscoz et al. 2005). Structures associated with roads such as culverts, low water bridges, and sewer lines may also impact invertebrate or fish communities by acting as dispersal barriers (Gibson et al. 2005; Wheeler et al. 2005). In New Zealand, road culverts acted as a partial barrier to upstream flight of adult aquatic invertebrates (Blakely et al. 2006). Overall fish movement was an order of magnitude lower through culverts versus other crossing types in Arkansas streams (Warren and Pardew 1998) probably due to the steep slopes and low hydraulic roughness that result in high water velocities (Belford and Gould 1989). In a study of sewer line crossings in Pennsylvania, the species richness, biomass of fish assemblages, and IBI scores dropped abruptly upstream (Koryak et al. 2001a).

Deicing salts from roads and airports are another source of pollutants to adjacent streams. There is little information on the effects to freshwater invertebrates. Laboratory studies showed that an amphipod and two species of caddisflies exposed to various concentrations of NaCl experienced increased drift and mortality in contrast to other tested invertebrates (Blasius and Merritt 2002), and a study of road deicers in Pennsylvania reported shock loads of salt that were sufficiently elevated to cause osmoregulatory stress to freshwater biota (Koryak et al. 2001b).

Major airports typically collect stormwater runoff and direct it to water treatment facilities. However during extreme storm events runoff may be diverted directly into an associated stream. In the winter this may lead to the introduction of ethylene or propylene glycol deicers. A study near Stapleton airport in Denver determined that there was no significant impact after the introduction of airplane deicers on the benthic macroinvertebrate and fish communities in a nearby creek that had received large volume of these deicers (Pillard 1996). However, it should be noted that the community was all ready seriously degraded throughout the reach. In contrast, an impacted stream near an airport in Pennsylvania had a severely stressed invertebrate community and an impaired fish community that the investigators attributed to a combination of high BOD demand from microbial metabolism of deicing agents and high ammonia concentrations from the breakdown of associated urea (Koryak et al. 1998).

Chemicals introduced by road usage may have negative effects on aquatic communities depending on the amount of traffic, the diluting effects of the rainfall and the size of the stream. Studies of macroinvertebrate communities in streams in the United Kingdom that are heavily impacted by highway runoff have shown reductions in pollution-intolerant species such as mayflies, stoneflies, and caddisflies (Maltby et al. 1995; Beasley and Kneale 2003). Benthic species are believed to be most at risk due to their exposure to both contaminated water and sediments. In Virginia, a study of the whole body dry weight of impacted benthic insects and fish found that lead concentrations were significantly correlated with traffic density (Van Hassel 1980), and bioaccumulation of automotive catalysts such as platinum and rhodium has been shown in an aquatic isopod (Rauch and Morrison 1999) and fish (Ek et al. 2004). Although no reports were found in the literature, it would be reasonable to assume that any airport runoff would lead to the introduction of many of the compounds associated with road usage and possibly result in the same sorts of impacts to the aquatic biota.

Not surprisingly, chemicals introduced into rivers by rail car spills have had serious effects. In a 1992 train accident, a complex mixture of aromatic hydrocarbons was released into a tributary of Lake Superior. The result was a major fish kill along with exposure for several weeks to an inestimable number of fishes. Documented effects included lamellar fusion and basal hyperplasia (Caldwell 1997).

Thresholds

In a review of highway impacts by Wheeler and others (2005), the authors suggested that urban development had more severe effects on impacted streams due to the fact that paved roads contribute only a fraction of the total impervious surface to an urban watershed. In contrast, authors of a study of roads in Alaska reported that storm drains and roads were important elements influencing the degradation of water quality and the biota (Ourso and Frenzel 2003). Finally, a study that ranked the impact of various factors on streams (Mattson et al. 2007) ranked bridges as having a high impact on water and habitat quality, and a medium impact on biota, flow, and energy sources. Road density was ranked as having a high impact on water and habitat quality, and a low impact on the biota, flow, and energy sources; while railroad density was ranked as having a medium impact on water and habitat quality, and a low impact on the biota, flow, and energy sources.

Urbanization

Recently the Environmental Protection Agency reported the results of the assessment of over 23% of the nation's rivers and streams. Of these waterways over 35% were listed as impaired with more than 80,000 miles impacted by urban land uses such as runoff and storm sewers, municipal point sources, and land disposal (USEPA 2000). In large watersheds urban land usually composes only a small percentage of the total area, but percentages may be greater than 90% in small watersheds (Wang et al. 2001). In both situations urbanization has been shown to have a disproportionate effect on the associated rivers and streams (Paul and Meyer 2001; Allan 2004). Impacts may be caused directly by channel modification, riparian clearing, water withdrawal and introduced species, or indirectly through changing land use resulting in increased impervious surface and stormwater runoff (Brasher 2003; Booth et al. 2004) with the level of response influenced by the slope, storage capacity, conveyance and connectivity of the watershed, and the inherent stability, erodibility, and riparian condition of the stream channel itself (Bledsoe and Watson 2001).

General agreement exists that the most serious impact is stormwater runoff which results in a flashier hydrograph, altered channel morphology, elevated levels of nutrients and pollutants and reduced biological diversity (Walsh et al. 2005; van Duin and Garcia 2006). Other changes may include increased water temperature due to loss of riparian vegetation and warm surface runoff, bank erosion and instability, loss of instream habitat from increasing sediment inputs, and a general loss of interaction between the stream and its watershed (Allan 2004). Research has shown a variety of urban indicators that have been correlated with these changes including impervious surface cover, housing density, and human population density (Klein 1979; Schueler 1994a; May et al. 1997; (ERM) 2000; Brown 2000). However, other studies have suggested that riparian conditions such as width, length and connectivity, along with distance to the land use patch may affect the relative impact of these land use indicators (Schuft et al. 1999; Stewart et al. 2001; Wang et al. 2003; McBride and Booth 2005; Snyder et al. 2005).

As prime agricultural land is urbanized, increasing amounts of marginal farmland are put into production. This can have significant water quality consequences because marginal lands are generally steeper, have more erodible soils, poorer drainage, and require more fertilizer than prime farmlands (Charbonneau and Kondolf 1993). Another area of concern is the tendency for urban stream banks to be straightened, narrowed, and paved leaving them more subject to erosion. These streams can develop the same problems as channelized, agricultural streams (Waters 1995).

Recommended approaches to dealing with the effects of urbanization on watersheds include land use planning, street cleaning, erosion controls, percolation ponds, and detention/sedimentation basins, with the major emphasis on source storage. Structural measures include in-line and off-line storage, sedimentation and biological treatment lagoons to control combined sewer overflows (Finnemore and Lynard 1982). Others are developing new approaches and methods of evaluation that can be used to identify where and how land use decisions may have critical influence over environmental quality, allowing planners to focus on future research and monitoring efforts, and other watershed protection measures (Naiman et al. 1993; Lee et al.

1994; Sutherland 1995; Naiman 1996; Rapport et al. 1998; Wear et al. 1998; Dale et al. 2000; Scholz and Booth 2001).

Flow regime

Increasing impervious surface in a watershed interferes with the natural movement of water within the system. Precipitation that would normally return to the atmosphere or slowly percolate down through the soil recharging the groundwater is instead routed overland directly into the channel along with fine sediment and other pollutants that are picked up along the way (Meyer and Wallace 2001). The loss of groundwater reduces the length of the base flow for habitat maintenance (Power et al. 1999), while the increase in runoff into a receiving stream or river results in higher peak discharge, flood frequency, and total runoff, while flood duration and lag time between rainfall and runoff decrease (Konrad et al. 2005; Chin 2006; Poff et al. 2006). In comparison to natural groundcover, watersheds with 10 – 20% impervious surface may produce twice as much runoff, those with 35 – 50% impervious surface may produce three times as much runoff and impervious surface cover of 75 – 100% within a watershed may produce more than five times as much runoff (USEPA 1993). In addition, Klein (1979) reported that Leopold stated that there was a direct relationship between the extent of urbanization within a watershed and the number of annual bankfull flows. For gravel bed streams the resulting flashier hydrograph interferes with its geomorphic equilibrium making it more difficult for the streams to adjust and compensate for the changes (Konrad and Booth 2005).

Another source of significant change in substrate and flow conditions may result from industrial effluents. A study of surface waters in Michigan reported that industrial effluent increased total stream discharge by 50-150% (Nedeau et al. 2003). Increased variability in stream flow impacts the biotic communities which are adjusted to the natural fluctuations of the system (Morley and Karr 2002). While some studies have shown that reduced groundwater recharge leads to lower median and minimum daily flows (Klein 1979) others have reported the opposite (White and Greer 2006). These contradictory results are likely due to inputs to some streams from leaking septic systems and wastewater treatment plants that supplant groundwater losses (Paul and Meyer 2001; Burns et al. 2005). Water withdrawals may complicate this situation further by reducing base flow (Waters 1995).

Physical habitat

Urban development impacts associated streams both directly and indirectly. Urban streams are often subjected to intense modification by developers. Small headwater streams may be filled in without regard to the ecological consequences (Meyer and Wallace 2001), while larger stream channels may be straightened, narrowed and even paved in an effort to contain flow (Waters 1995). Unaltered headwaters not only reduce the risk of flooding, they also reduce the occurrence of drought-like conditions downstream. When small streams are maintained in their natural state they are capable of absorbing significant amounts of rainfall and runoff prior to flooding (Meyer and Wallace 2001). Channel modifications to larger streams in an attempt to control flooding or modify stream alignment to construction may impact several physical features of the associated stream including its depth and stage, water surface area, channel morphology, current velocity, turbidity, and substrate (Stern and Stern 1980). Stream channels of all sizes, along with their riparian corridors and flood plains retain both water and fine sediments that reduce erosion and sedimentation downstream (Meyer and Wallace 2001).

During the construction phase, sediment supply usually increases leading to bed aggradation and overbank deposition (Paul and Meyer 2001). Cordery (1976) reported that during construction of urban developments, sediment production in associated streams was increased by a factor of 50 or more. In Michigan, a five-fold increase in suspended sediment was reported in a stream where the banks were left unprotected after nearby construction (Hansen 1971). Construction along a stream in southern Ontario, resulted in an increase in suspended solids to more than 1000 mg/L as compared to normal levels of < 5 mg/L, and the unprotected banks continued to erode at an accelerated rate after the completion of construction (Barton 1977).

After construction ends the sediment supply is reduced, but the loss of small headwaters along with extensive vegetative removal and soil compaction produces increased runoff which leads to increased hydraulic force acting on the receiving stream channel. Increased stream power from urbanization may result in increasing turbidity, increasing cross-sectional area of the channel (width, depth and width/depth ratio), and increasing large woody debris (LWD) recruitment due to bank instability and erosion, while causing decreases in cover, and LWD retention (Finkenbine et al. 2000; Doll et al. 2002; Snyder et al. 2003; McBride and Booth 2005; Chin 2006). The drop in quantities of LWD results in a loss of pool habitat and channel complexity (MacDonald et al. 1991; Hilderbrand et al. 1997; Gomi et al. 2002) and an even greater increase in stream power since LWD in the channel slows down flow and reduces stream power. Alterations in the bed sediments seem to be the least predictable physical change. Studies in Washington and Georgia reported a loss of relative roughness, a reduction of the median particle size (Morley and Karr 2002; Roy et al. 2003; Walters et al. 2003) and increasing cementation (McBride and Booth 2005). However, Chin (2006) who reviewed more than a 100 studies on the effects of urbanization on streams reported that in general physical changes to bed sediments resulted in coarser bed material due to scouring of fines—although she acknowledged that sometimes large quantities of sand remained. This concurs with a study of the effects of urbanization in Maryland that reported urbanizing areas not only generated 9 times more sediment as rural drainages, but that the total volume of sand generated could be as much as 15 times that found in nearby rural streams (Fox 1974). Due to its instability, sand is the poorest substrate for benthic species.

Energy sources and relation

Urbanization has many of the same impacts as agriculture on associated streams because it too alters the two primary energy sources: terrestrial material and material that is fixed within the stream (Young and Huryn 1999). The ratio of these two carbon sources likely determines overall productivity of the stream, and hence, the composition of the resident invertebrate and fish assemblages. Removal of the vegetation in the watershed and riparian areas, along with soil compaction reduces the terrestrial inputs such as leaves and woody debris, while adding large quantities of fine sediment. This reduction of allochthonous inputs results in less availability of fine particulate organic matter for downstream food webs and reduced secondary production (Wallace et al. 1999; Meyer and Wallace 2001), while increased sediment loads may impact the physiology, feeding or reproductive behaviors of fish and invertebrates (Buck 1956; MacLeod and Smith 1966; Aldridge et al. 1987; McLeay et al. 1987; Sweeten and McCreedy 2002). Woody debris dams within the channel act as retentive structures. When these are unavailable the export and turnover of benthic particulate organic matter is accelerated (Webster et al. 1990). Removal of riparian canopy also results in increased water temperature and light penetration that increases autochthonous production (Cummins 1974; Pinto et al. 2006). These increases may be somewhat lessened by increased turbidity (USEPA 1986; Lloyd et al. 1987). Other impacts of vegetative removal and soil compaction include less groundwater flow. Groundwater modulates water temperature while supplying oxygen and dissolved inorganic and organic nutrients that promote stream productivity (Power et al. 1999).

Several studies have shown that various ecosystem functions are affected by increasing urbanization. Studies in catchments in Georgia showed that ecosystem functions of the stream such as uptake velocities of NH_4 and soluble reactive P decreased along with increasing urbanization (Meyer et al. 2005), as did average organic matter standing stocks (Paul 1999), and fungal biomass on decomposing leaves (Paul et al. 2006). In Oregon, a study found that diatom assemblages responded to impacts on urban streams (Walker and Pan 2006), and gross primary production and community respiration were higher in urban streams in Georgia (Paul 1999). Urbanization has also been reported to affect nitrogen retention in New Hampshire via changes in nitrogen loading and processing (Wollheim et al. 2005), while a shift in organic C bioavailability was found in Australia (Harbott and Grace 2005).

Specific features of urbanization and associated impacts that have been reported on include relations between stormwater pipes and algal biomass (Taylor et al. 2004), waste water treatment plant (WWTP) effluent or CSO discharge and the stable isotope signatures of fish and macroinvertebrates (Northington and Hershey 2006), WWTP effluent and the composition of suspended fine particulate organic matter (Rosi-Marshall 2004), and WWTP effluent and bioavailable phosphorous (Mainstone and Parr 2002).

Water quality

Watersheds concentrate runoff, and in this process, they are shaped by the transport and production of water and sediment. In turn, these media transport other materials, all of which have an effect on the water chemistry within the watershed. Alteration of land use by human activities directly impinges on these processes (Reid 1993). Runoff from developing urban settlements, as well as existing urban structures, has been found to constitute a major source of pollution for rivers. Research on an urban stream in Pennsylvania indicated that urban surface runoff should be regarded with equal concern to other wastewaters generated by human activities (Brush et al. 1979). Chemical effects of urbanization on surface waters depend on the extent and type of urbanization (residential/commercial/industrial), presence of WWTP effluent and/or CSOs, and the handling of stormwater drainage. Those factors that most often increase are oxygen demand, conductivity, suspended solids, ammonium, hydrocarbons, and metals. Phosphorous, ammonium, and nitrogen concentrations may also be elevated in urban streams (Berndtsson et al. 1989; Wernick et al. 1998; Fisher et al. 2000; Paul and Meyer 2001; Snyder et al. 2003; Grapentine et al. 2004; Brett et al. 2005; Clinton and Vose 2006), although (Ahearn et al. 2005) reported that stream nitrate levels were not impacted by human habitation until a WWTP was built within the watershed.

The results from a 1.5-year study of the groundwater quality near Milwaukee, Wisconsin, suggested that chloride and sulfate were the principal products of urbanization that altered the groundwater, due to infiltration of polluted surface waters. In addition, relatively high concentrations of fecal coliform and fecal streptococci bacteria were often found (Eisen and Anderson 1979). Chloride can be used as a biological indicator of human waste within water samples (Herlihy et al. 1998), but is not of particular concern to freshwater quality except at very high levels. However, elevated levels of fecal coliform may be an indicator of potentially serious public health problems. Determining the source of these bacteria can be complicated by mixed uses of a watershed, such as animal production, population centers, and recreational use. Recently, methods of identifying host sources of fecal coliform in water have been developed to assist in the formulation of pollution reduction plans (Carson et al. 2001; Murray et al. 2001).

An analysis of demographic and land-use factors done on rivers within coastal estuaries demonstrated that fecal coliform abundance was significantly correlated with watershed population, and even more strongly correlated with the percentage of developed land within the watershed. However, the strongest correlation was between fecal coliform abundance and percentage of watershed with impervious surface coverage, such as roofs, roads, driveways, sidewalks, and parking lots (Mallin et al. 2000). Other studies have linked increased bacterial loads with two common effects of urbanization: increased suspended sediments and high flows (Elder 1987; Irvine and Pettibone 1996; Baudart et al. 2000). Bacteria appear to be related to turbidity and suspended sediment, suggesting that solids may play a role in transporting bacteria (Sowitzki et al. 1996; Baudart et al. 2000; Murray et al. 2001).

Other pollutants associated with urban runoff include chemicals from home use, commercial/industrial use, and lawn and golf course applications (Line et al. 1997; Bailey et al. 2000; Hoffman et al. 2000; Paul and Meyer 2001; Kolpin et al. 2006). Pesticide application rates for golf courses have been reported to significantly exceed those for common agricultural crops (Schueler 1994b). A study in France suggested that suburban uses of herbicides might endanger drinking water derived from river water (Blanchoud et al. 2004), while Bailey and others (2000) reported that pesticide concentrations in California streams were higher in a watershed receiving mostly residential inputs versus a watershed largely impacted by commercial and industrial activities. Finally, a study of insecticide use in urban areas of the United States found that two commonly used household insecticides, carbaryl and diazinon, exceeded criteria for the protection of aquatic life in many urban streams during the growing season (Hoffman et al. 2000).

Other compounds that are found in stream water and sediments include metals such as lead, zinc, chromium, copper, manganese, nickel and cadmium (Good 1993; Lenat and Crawford 1994; Line et al. 1997; Duke et al. 1998). High iron and zinc concentrations were reported in streams impacted by leachate from an Ontario landfill (Dickman and Rygiel 1998), while a study in Texas reported rooftop materials including asphalt shingles and metal roofing were major contributors to streams of zinc, lead, pyrene, chrysene, chromium, and arsenic (Van Metre and Mahler 2003). A study of sewage sludge application to dairy fields showed that total streamwater

export of Cu, Na, Mo, and soluble P were greater from the biosolids watershed than from the watershed treated with fertilizer (Richards et al. 2004). A whole suite of organic contaminants are frequently detected in urban streams including polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and aliphatic hydrocarbons (Paul and Meyer 2001) with high levels of PAHs reported from oil refinery effluent in Delaware (Uhler et al. 2005). A study of CSO outfalls along the Passaic River in New Jersey suggested that they were the primary source in associated sediments of a wide range of chemicals including toxic metals, PAH, PCB, pesticides, and other organic chemicals (Iannuzzi et al. 1997).
Perhaps the most disturbing reported contaminants from surface waters are pathogenic protozoa (Gibson et al. 1998; Stadterman et al. 1998); estrogens (Gagne et al. 2001); and a wide variety of pharmaceuticals, hormones, and other organic wastewater contaminants including coprostanol (fecal steroid), cholesterol, DEET, caffeine, triclosan, tri(2-chloroethyl)phosphate (fire retardant) (Kolpin et al. 2002), ibuprofen, naproxen, carbamazepine, and acetaminophen (Brun et al. 2006). Pollutants may display one of the following patterns in samples collected immediately after storm events: decreasing concentrations due to dilution by storm-water runoff, increasing concentrations due to sediment input and/or surface runoff, or increasing concentrations attributed to storm interflow (Klarer and Milie 1989).

Urban development may also alter stream temperature, potentially limiting thermal habitat that may lead to changes in fish community structure (Krause et al. 2004). A study in Michigan found that industrial effluent affected water quality by raising in-stream temperatures 13-18 °C during colder months and carrying high amounts of iron (> 20 x higher than ambient) that covered the streambed (Nedeau et al. 2003). Effluents produced by pulp mills and sewage plants have been shown to reduce levels of dissolved oxygen in the substratum (Lowell and Culp 1999), while combined sewer overflows reduce dissolved oxygen during storms flows causing fish kills (Chen et al. 2004).

Biotic interactions

Urbanization alters the flow regime, physical habitat, energy sources and water quality of associated streams. This leads to impairment of the aquatic biota (Paul and Meyer 2001; Allan 2004). Studies of the impacts on aquatic invertebrates have found that urbanization results in decreased richness, diversity and overall abundance of pollution intolerant taxa, while increasing the relative abundance of pollution tolerant taxa (Lenat and Crawford 1994; Sponseller et al. 2001; Moore and Palmer 2005). Similar results have been reported for urban fish communities, along with increases in introduced species (Wang et al. 2000; Brasher 2003; Snyder et al. 2003; Rashleigh 2004; Schweizer and Matlack 2005; Roy et al. 2006).

Most studies looking at the effects of urbanization on streams have simply linked the amount of impervious surface to various biological measures of degradation (Klein 1979; Roth et al. 1996; Hicks and Larson 1997; Baker and Sharp 1998; Maxted and Shaver 1998; (ERM) 2000; Brown 2000; Wang et al. 2000; Wang et al. 2001; Morse et al. 2003; Weber and Bannerman 2004; Moore and Palmer 2005). This makes it difficult to assess the exact relations to any particular variable since these bioindicators may be responding to a variety of interrelated effects including reduced infiltration and groundwater, increased runoff and erosion, increased severity of flooding and droughts, loss of channel sinuosity and riparian cover, increased water temperatures, and increased pathogens, nutrients, or toxicants.

Several studies have reported significant relations between various biological measures of macroinvertebrates and urbanization. Urban land use in Wisconsin had stronger relations to invertebrate measures than channel morphology, substrate or water quality (Wang and Kanehl 2003). In Virginia, (Jones and Clark 1987) reported that invertebrates in the relatively pollution tolerant order Diptera composed 12 – 36% of the invertebrate community at rural sites, 14 – 66% at moderately urbanized sites, and 33 – 99 % at heavily urbanized sites. In North Carolina, relative abundance of EPT taxa was an order of magnitude lower in an urban stream than in either a forested or agricultural stream (Lenat and Crawford 1994), and similar results were found in Wisconsin (Stepenuck et al. 2002) and Tennessee (Freeman and Schorr 2004). In Maryland, Klein (1979) reported reduced invertebrate richness with increasing urbanization, as did others in later studies (Strayer et al. 2003; Moore and Palmer 2005), while reductions in invertebrate diversity have been reported in Maryland (Schueler 1994a), Utah (Gray 2004), Connecticut (Urban et al. 2006), and Wisconsin (Wang and Kanehl 2003). Invertebrate IBIs have been reported to decrease with increasing impervious surface in the Pacific Northwest (May et al. 1997; Morley and Karr 2002).

Changes in fish communities have also been linked with urban development. Lower IBI values and species richness have been related to various measures of urbanization in Wisconsin (Wang et al. 2001; Wang et al. 2003) and an increased proportion of exotic fish species has been reported in Maryland (Strayer et al. 2003). Mobile lab studies of fathead minnow behaviors found that time spent conducting nest and/or egg care activities, average spawning attempts, and development of male secondary sexual characteristics were significantly related to the percentage of impervious surface in the watershed (Weber and Bannerman 2004).

Some studies have reported on relations between the biota and specific environmental changes due to urbanization. Density of filter feeding and grazing taxa, as well as that of EPT taxa, declined as a result of significant water withdrawals in a Michigan stream (Wills et al. 2006), and in Georgia, water withdrawal rate along with drainage area accounted for 70% of the among-site variance in fish generalist and specialist species, and were better predictor variables than percent of urban land use or average sediment size (Freeman and Marcinek 2006). Heavily engineered sites in an English river had lower quality macroinvertebrate fauna than did less modified sites (Beavan et al. 2001), as did the discharge area of a WWTP in France (Montuelle et al. 1997), the discharge areas of surface water outfalls on a river in England (Robson et al. 2006), a stream associated with a landfill in Ontario (Dickman and Rygiel 1998), landfill leachate discharge in Canada (Rutherford et al. 2000), an abandoned landfill in Montana (Marshall 2001), and toxic releases (hexachlorobutadiene) in Canada (Taylor et al. 2003). WWTP effluent has also been reported to negatively affect fish diversity and abundance in an English river (Harkness 1982).

Work in Georgia on macroinvertebrates (Roy et al. 2003) found that biotic indices were predicted better by reach-scale variables such as sediment size, TSS and water chemistry than land cover variables. Other work by the same group suggested that hydrologic variables explained 22 to 66% of the variation in fish richness and abundance (Roy et al. 2005), and that abundances of a federally threatened darter were best predicted by models with single variables representing stormflow and sediment conditions (Roy et al. 2006). Other studies in Georgia, reported that urban development disrupted the relationships between fishes with both sediment and slope, favoring cosmopolitan species (Walters et al. 2003); and that increased turbidity from urban runoff increased the number of fishes with deformities and had a negative influence on the number of sensitive species, number of individuals, proportion of lithophilic spawners, and proportion of omnivores (Schleiger 2000).

Hazardous waste sites have been incriminated in declining frog populations in New York (Quimby et al. 2005), and toxic contamination of fishes in Virginia (Pinkney and McGowan 2006) and snapping turtles in Canada (de Solla et al. 2001). Biomagnification of PCBs within the entire food web has been reported from the Kalamazoo River superfund site in Michigan (Blankenship et al. 2005; Kay et al. 2005), and feeding studies with contaminated diet collected from the Clark River superfund site in Montana indicated that trout are at high risk from elevated metals concentrations in surface water, sediment, and aquatic invertebrates (Pascoe et al. 1994). A study of the Clinch River, Tennessee, where three U.S. Department of Energy facilities reside showed that fish exposed to associated sediments with the highest levels of mercury and PCBs, exhibited organ dysfunction, increased frequency of histopathological lesions, impaired reproduction, and reduced fish community integrity (Adams et al. 1999). No reports of impacts to aquatic biota from power lines were found, but regulations and management plans for construction activities in sensitive riverways are in place (PB 2002). Possible impacts might include those discussed above resulting from urban development.

Thresholds

Table 1. Reported threshold levels of urbanization that may impact aquatic biota.

Study subjects	State/Province	Threshold	Response	Source
Invertebrates/ISC[1]	MD	>15%	Diversity decreased	Schueler 1994a
Invertebrates/ISC[2]	DE	8–15%	Diversity decreased	Shaver et al. 1995
Invertebrates/ISC[2]	DE	10-15%	Loss of sensitive taxa	Maxted 1996
Invertebrates/ISC[2]	WA	5–10%	Metric scores decreased	May 1997
Invertebrates/ISC	WI	>8–12%	Diversity & EPT richness decreased	Stepenuck et al. 2002
Invertebrates/ISC	ME	>6%	Richness decreased	Morse et al. 2003
Invertebrates/ISC	WI, coldwater	<7%	EPT richness & abundance decreased	Wang & Kanehl 2003
Invertebrates/fish/ISC	MD	<6% ISC + 65% tree in riparian	Excellent stream health rating	Goetz et al. 2003
Invertebrates/fish/ISC	MD	<10% ISC + 60% tree in riparian	Good stream health rating	Goetz et al. 2003
Fish/ISC	MD	>12%	Diversity decreased	Klein 1979
Fish/ISC	MD, sensitive streams	>10%	Abundance decreased	Klein 1979
Fish/ISC	MD	30–50%	Fish absent	Klein 1979
Fish/ISC	MD	>10–12%	Diversity decreased	Schueler 1994
Fish/ISC	WI	>10-20%	IBI decreased	Wang et al. 1997
Fish/ISC	OH	>8%	IBI decreased	Yoder et al. 1999
Fish/ISC	OH, watersheds < 100 mi^2	>15%	IBI decreased	Yoder et al. 1999
Fish/ISC	WI	>10%	Richness & IBI variable	Wang et al. 2000
Fish/ISC	WI	>8-12%	Richness & IBI decreased	Wang et al. 2001
Fish/ISC	WI, coldwater	< 6%	High IBI	Wang et al. 2003
Fish/ISC	WI, coldwater	>11%	Low IBI	Wang et al. 2003
Fish/urban land	WV	>7%	Low IBI	Snyder et al. 2003
Fish/urban land	MD	>40-50%	>80% failing IBI value	Volstad et al. 2003
Fish/urban land	OH	>13.8%	IBI decreased	Miltner et al. 2004
Fish/urban land	MD	>25%	Low abundance & richness	Morgan and Cushman 2005
Fish/urban land	MD	<20%	Fish abundance - reference	Stranko et al. 2005
Fish/urban land	MD	>50%	Fish abundance - degraded	Stranko et al. 2005
Amphibians/urban land	CA	>8%	Abundance & richness decreased	Riley et al. 2005

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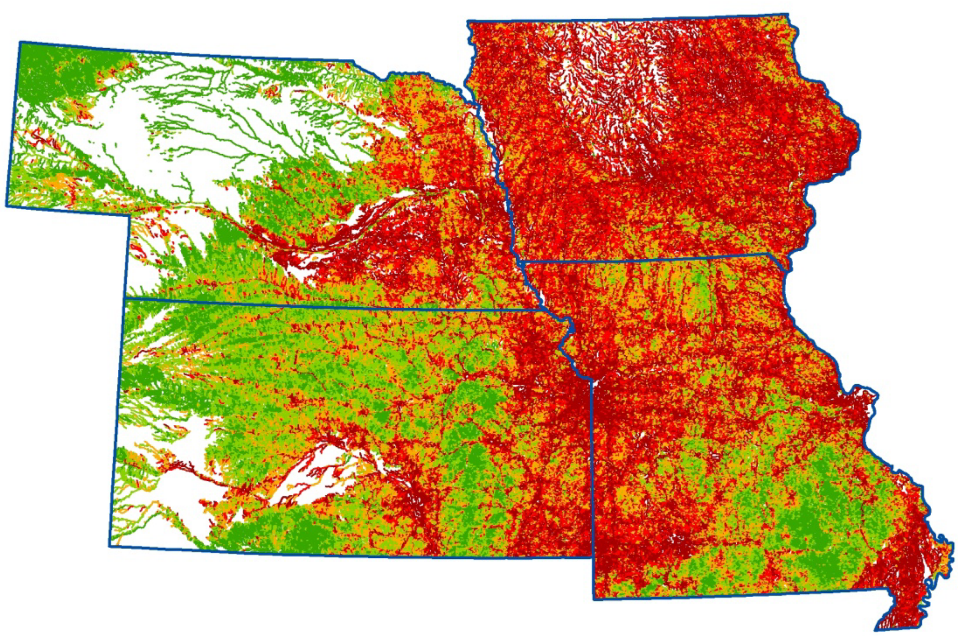
Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7

Missouri Department of Natural Resources

Appendix F

Literature Review of Methods for Assessing Cumulative Impacts



Literature Review for GIS-based Methods for Assessing Cumulative Impacts and Overall Environmental Quality

When using GIS-based methods to evaluate overall environmental quality of riverine ecosystems, the most common practice is to consider the human stressors or threats to any given stream system. The primary variable researchers take into consideration is the land use/land cover of the surrounding area. Stream quality can be measured by looking closely at how the anthropogenic disturbances have affected the macroinvertebrate and fish communities by creating threat indices. Another variable researchers consider, oftentimes, is the scale. Frequently evaluations are conducted at 3 different scales, but can be run and rerun at as many scales or resolutions as desired. It is often the differences in the results at varying scales that reveal significant information during the study. The challenge for researchers is to develop a threat index that is correct at a catchment (or smaller) scale, but that can also be accurate at a regional level. Most studies begin by assessing the land use and other human stressors in a watershed area.

Land use can be divided many different ways during GIS analysis depending on what study area is under review. The number of classes used is frequently at the discretion of the researcher. Agriculture, forest and urban classes often form the three basic classes (Allan 1997, Moore 2005, Rashleigh 2004, Van Sickle 2004). Study areas can be limited to one land use class, such as an inner city urban area (Booth 2004, Taylor 2004), or can require a researcher to implement Anderson Level II land use classification (Host 2005), but this is rare. Many studies use somewhere between five and ten land use classes, often depending on the scale of the project (Arbuckle 2002, Bruns 2005, Diamond 2002, Locke 2006, McMahon 2000, Richards 1996, Roth 1996, Rothrock 1998, Schleiger 2000, Stepenuck 2002, Strayer 2003, Whiles 2000). Any number of anthropogenic disturbances, besides land use, can be incorporated into an environmental quality study. Wang (2008) considered population density, transportation, nutrient enrichment, agricultural pollutants, and point source pollution. McAbee's (2008) data included mining permit polygons, point locations of oil and gas wells, locations of logging inspection sites, percent of watershed under mining permits, percent of watershed urbanized, percent of watershed in pasture, number of logging inspection sites in watersheds, well density and road crossings over streams. There are any number of variables that can, and oftentimes should, be considered in an environmental quality study. The number of variables are limited by data availability, time or money, but not by GIS capabilities.

Once researchers have established the threat index they need a determinant for stream quality. Macroinvertebrate communities are often used as indicators of stream health because of their varying "life histories that are sensitive to degradation" (Wang 2003). Fish sampling is also used to gauge stream quality. Once sampling is complete the biotic data is, often, developed into an index of biotic integrity (IBI). It is at this point that studies diverge as there are many methods of statistical analysis with which to process the indices.

Some studies choose to weight variables using localized inverse-distance weighting (Kapo 2006). Some chose to calculate their indices based on deviation from a reference condition, or baseline (Karr 1991, Booth 2004, Norris 2007, Wang 2008). Some studies weighted variables based, seemingly, more on the opinion of the researchers (Bryce 1999). Once the variables are weighted, or assigned a risk index score (Bryce 1999), the IBIs are correlated with the land cover, or other indices, at the desired scale(s) to produce a cumulative index (Sponseller 2001, Morley 2002, Stepenuck 2002, Wang 2003). Results vary depending on scale.

The strength and flexibility of GIS-based methodology begins to show itself when the discussion turns to spatial resolution, or scale. The ability to quickly run and rerun combinations of variables at different scales saves time and money. Many researchers prefer to run their models at three scales (Booth 2004, Bressler 2006, Griffith 2000, Lammert 1999, McMahon 2000, Morley 2002, Rios 2006, Roth 1996, Snyder 2003, Sonoda 2001, Strayer 2003, Sutherland 2002, Townsend 2003, Van Sickle 2004, Whiles 2000). The smallest scale is usually a riparian corridor along the stream. The mid scale is usually at catchment or segmentshed scale. The largest scale is often run at the watershed level. The occasional study will look at a regional scale (Danz 2007). The challenge for researchers is to build a cumulative index that will be applicable at both small-scale, riparian reaches, and large-scale, regional reaches.

Some studies conclude that their indices show stronger correlations between anthropogenic disturbances and macroinvertebrate decline at one scale than the others, usually a smaller, riparian scale (Sponseller 2001, Whiles 2000). Rios (2006) asserts that “riparian-reach variables influence” macroinvertebrates more than catchment scale land use variables. Booth (2004) found significant correlations between urban land cover and benthic IBI at all three spatial scales used in their study. Stepenuck (2002) concluded that “urbanization significantly degraded stream macroinvertebrate communities, hence stream quality”. Several researchers agree there appears to be a threshold level of watershed imperviousness of approximately 7-8%, that once crossed results in significant declines in macroinvertebrate and stream health (Snyder 2003, Stepenuck 2002, Wang 2003)

The use of fish sampling to evaluate impacts of human threats is another method frequently used by researchers. Snyder (2003) used two different indices of biotic integrity when evaluating the effects of land use on their study area. One of their conclusions was “assigning fish species to broad ecological categories based on the literature is a subjective process” (Snyder 2003). Statements like this demonstrate the problems researchers have with fitting individual fish species into rigid ecological classifications that will ultimately be used to calculate a metric score.

There are many ways of classifying land use and human impacts on the environment. Macroinvertebrates are an excellent indicator of stream health because of their varying life spans and sensitivity to changes in stream ecosystems. Evaluations of stream quality can be run at many different scales to ensure accurate representations of the state of the watershed. Due to the diversity and mobility among fish species, researchers generally find macroinvertebrates metrics easier to correlate with land use metrics. A threshold level of 7-8% impervious surface in a watershed is all that is required to created significant declines in the health of a stream.

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