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Analysis

An Evaluation of Proposed New Park Units in Idaho Using GAP Analysis Data

Conservation groups have proposed four new areas of Idaho to be administered by the National Park Service. The areas encompass an average of 220,000 h. and contain important scenic, recreational, and geological resources. The biological resources which would be protected by these proposals, however, have received little comparative regional evaluation. This study used the U.S. Fish and Wildlife Service's Gap Analysis databases for Idaho to evaluate the vegetal resources contained in each proposal and within each ecoregion as they occur in Idaho only. Databases of vegetation, land ownership, and land management, were displayed in map format and analyzed using Arc/Info. Vegetation types were used as surrogates for information on the distribution of other biological resources. A conservation strategy was evaluated that would preserve at least 10% of the land area of each vegetation type in proportion to its occurrence.

Although an average of 85% of the lands in the three ecoregions in Idaho are publicly owned, only 15% of the state's vegetation types are adequately managed for their long-term persistence. The four proposals added little to this figure because the majority of the vegetation resources (67-78%) found in the proposal areas were already protected elsewhere in the state. We analyzed the effect of modifying the boundaries of the proposals to enhance biodiversity conservation. None of the proposals could be feasibly modified to fully conserve all of the vegetation types in the state. However, the potential conservation provided by all of the proposals could be enhanced substantially with the addition of relatively small areas of land identified as gaps.

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Steps in Strategies to Manage Biodiversity: Identification, Selection, and Design of Special Management Areas

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Gap Analysis provides a regional perspective on the distribution of several elements of biodiversity, notably, plant communities and vertebrate species. The maintenance of much biodiversity will depend on balanced management of multiple-use wildlands. Special management areas however, are a necessary component of an overall biodiversity management strategy, since they serve as a haven for those species and communities incompatible with multiple use management and provide control areas to assess the success of various management prescriptions outside of special management areas.

In their 1994 book, Saving Nature's Legacy, Reed Noss and Allen Cooperrider conclude, "The United States has no national strategy to conserve biodiversity." Aside from the opportunistic protection of scenic wilderness, habitat protection in the USA largely has been focused on areas inhabited by game species or endangered species. Although the recovery needs of species on the brink of extinction are legitimate components of an overall strategy to maintain biological diversity, they must be complemented by a proactive approach to land use planning that ensures that the bulk of biodiversity never becomes endangered in the first place. In an ideal world, an objective consideration of the distribution of biodiversity would lead to the identification of priority areas which would then be managed for their natural values in order to minimize future anthropogenic extinctions. This, of course, has never been the case. In reality, most natural areas have been set aside because they have little economic value, because of their scenic appeal, and because the opportunity to designate them presented itself. The primary danger of opportunistic development of a special management area network is that options to establish new special management areas could be exhausted before all elements of biodiversity are represented in the special management area system.

Developing a natural area network is a multiple step process. First, the distribution of the known elements of biodiversity must be assessed. Next, a set of areas is identified in which all elements of biodiversity are represented. This is an exercise in applied biogeography. Then, potential natural areas are more intensively studied to determine their condition and the feasibility of special management area designation. Sites meeting criteria for natural areas are then chosen. This process is commonly referred to as special management area selection. Following special management area selection, the principles of conservation biology are applied to delineate natural area boundaries sufficient to maintain viable populations and ecosystem processes. This step is commonly referred to as special management area design and draws on the disciplines of ecology, population biology, hydrology, and natural areas management. The spatial questions involved in identifying natural area networks in which biodiversity will be completely represented should not be confused with the practical and biological questions that need to be addressed when designing individual natural areas for long term viability of their constituent biodiversity elements and processes.

This entire process is complicated because of our incomplete knowledge of the occurrence and abundance of the elements of biodiversity, as well as an incomplete understanding of ecological processes. Our lack of knowledge is basic. We do not even have names for all species. Although estimates vary, perhaps 90 % of the world's species are unnamed. It is only for some of the higher vertebrates (large mammals, birds) that we have reasonably complete record. For others, especially invertebrates, we have a much less complete list of species. When it comes to more detailed ecological studies, such as distribution, abundance, demographics, and habitat association, we are far more ignorant. The same is true for process. Thus, while ideally identification, selection, and design of special management area areas should be based on complete knowledge, we are hindered by our ignorance of taxonomy and ecology of the species and the ecological processes occurring in the systems in which they live. However, we must not use lack of complete information as an excuse not to act on what biologically defensible information we do have. If we fail to do so, we will lose much of what we have.

Special management area Identification

Rather than focusing on locations of rare species or difficult-to-classify landscapes, biodiversity can be most efficiently represented if maps of several biodiversity elements are examined in hierarchical manner. First, areas in which all plant communities are represented are identified, corresponding to the "coarse filter" approach of The Nature Conservancy. Then, species-rich areas that are most complementary to one another are identified. Finally, areas containing species still unrepresented are located, a "fine filter" that catches species not represented in areas identified by the "coarse filter" approach.

A subset of areas from a state or region in which all biodiversity elements are represented can be identified using one of a variety of stepwise algorithms. This approach to conservation planning has been most fully developed in Australia. One algorithm, called the "greedy heuristic," proceeds as follows: The presence of plant communities or species becomes an attribute of an area; areas with the largest number of attributes are identified, then areas with the largest number of attributes not already present in the previous choice are identified, and so on. This stepwise approach maximizes complementarity in each successive selection and results in the efficient selection of a special management area network. Since many areas will share biodiversity attributes, alternative choices usually exist at each step, leading to the identification of different configurations of special management area networks, any one of which would be completely representative. Of course, areas containing unique attributes must be included in all potential special management area networks. These areas are irreplaceable (i.e., they must be included in all networks).

Designing and managing natural areas for the long term persistence of species and communities are important but fundamentally different issues than selecting potential special management area networks.

No amount of management will maintain species or ecosystems not present in a natural area network in the first place. However, the presence of a species or natural community in an area implies nothing about the potential of the area to maintain that species or community.

Special management area Selection

Once potential areas containing target species or communities have been identified, further information about the quality of each area needs to be gathered and compared with the biological, physical, and spatial requirements for long term persistence of the target species or communities. There are many established protocols for sampling plant and animal populations, and the intensity of sampling necessary to select the best natural area has not been systematically investigated and is likely to differ between ecosystem types. In some cases, a rapid assessment by trained biologists will suffice, in others, multi-year sampling of a number of populations will be necessary.

Social and economic factors are often more critical than biological factors when selecting among a set of potential special management areas. Cost, community attitudes, and projected changes in human land use in surrounding areas all contribute to the selection process. Possible ways to integrate these factors into special management area selection are being explored by Gap Analysis Programs.

Special management area Design

Population, community, ecosystem, and landscape processes are all important factors in special management area design. Furthermore, beyond the physical and biological components of special management area design, the size and shape of a natural area have considerable relevance to practical details of special management area management. Four areas of special management area design become relevant after potential natural areas are selected: 1) minimum area requirements for viable populations; 2) community-level interactions; 3) patch dynamics and other ecosystem processes; and 4) interactions between special management area design and management.

1. Many initial discussions of nature special management area design centered on the viability requirements for populations of target species, including population dynamics, the effect of environmental variation, genetics, metapopulation structure, and the effects of habitat fragmentation. In simple terms, natural areas must be large enough and have a shape that will support viable populations of most animal and plant species for a relatively long period of time, usually at least 100 years. Population viability analysis (PVA) represents an effort to formalize estimates of population persistence, but rarely are sufficient data available for robust conclusions.

Habitat quality varies spatially for most species, resulting in source and sink populations that interact as a metapopulation which experiences local extinction and colonization events. Habitat heterogeneity tends to increase with area, suggesting that larger natural areas offer more patches of high quality habitat which can carry a species through periods of adverse environmental conditions. Edge effects may result in negative population growth rates near natural area boundaries. Many species will occur in natural areas only when sufficient interior habitat is present. Edge is minimized and interior maximized as special management area shape becomes more compact.

2. The maintenance of essential community-level interactions and processes is the second major special management area design consideration. At the most basic level, natural areas need to support trophic interactions between producers and consumers. Some exchange of energy and matter will occur between special management areas and surrounding areas, so boundary delineation should always consider the context of natural areas. Carnivores typically occur at lower densities than herbivores of equal body size and often play essential roles regulating herbivore density and diversity. Special management areas must therefore meet the spatial requirements of the most area-sensitive community member. Mutualistic relationships exist between many plants and their animal pollinators, including insects, birds, and bats. Insuring the continuation of community interactions, especially those involving keystone species, becomes a primary special management area design challenge.

3. The concept that natural areas represent eternal and unchanging examples of particular ecosystems is a widely held fallacy (Botkin 1992). Many ecosystems experience regular disturbances whose frequency and patch size is an integral part of ecosystem function. Disturbance events include fire, windstorms, floods, landslides, and volcanism. While some catastrophic events affect large areas, most disturbances are local and scattered throughout a landscape. Special management areas ideally include the "minimum dynamic area, the smallest area with a natural disturbance regime." Disturbances would then occur in a shifting mosaic pattern within a natural area, with various patches in different stages of succession. This arrangement would ensure that propagules for recolonization of disturbed areas are present on undisturbed portions of the special management area. In practice, ecosystem management activities (such as controlled burning) can be used to recreate a natural mixture of seral stages on a smaller scale where natural disturbance events are larger than the natural area.

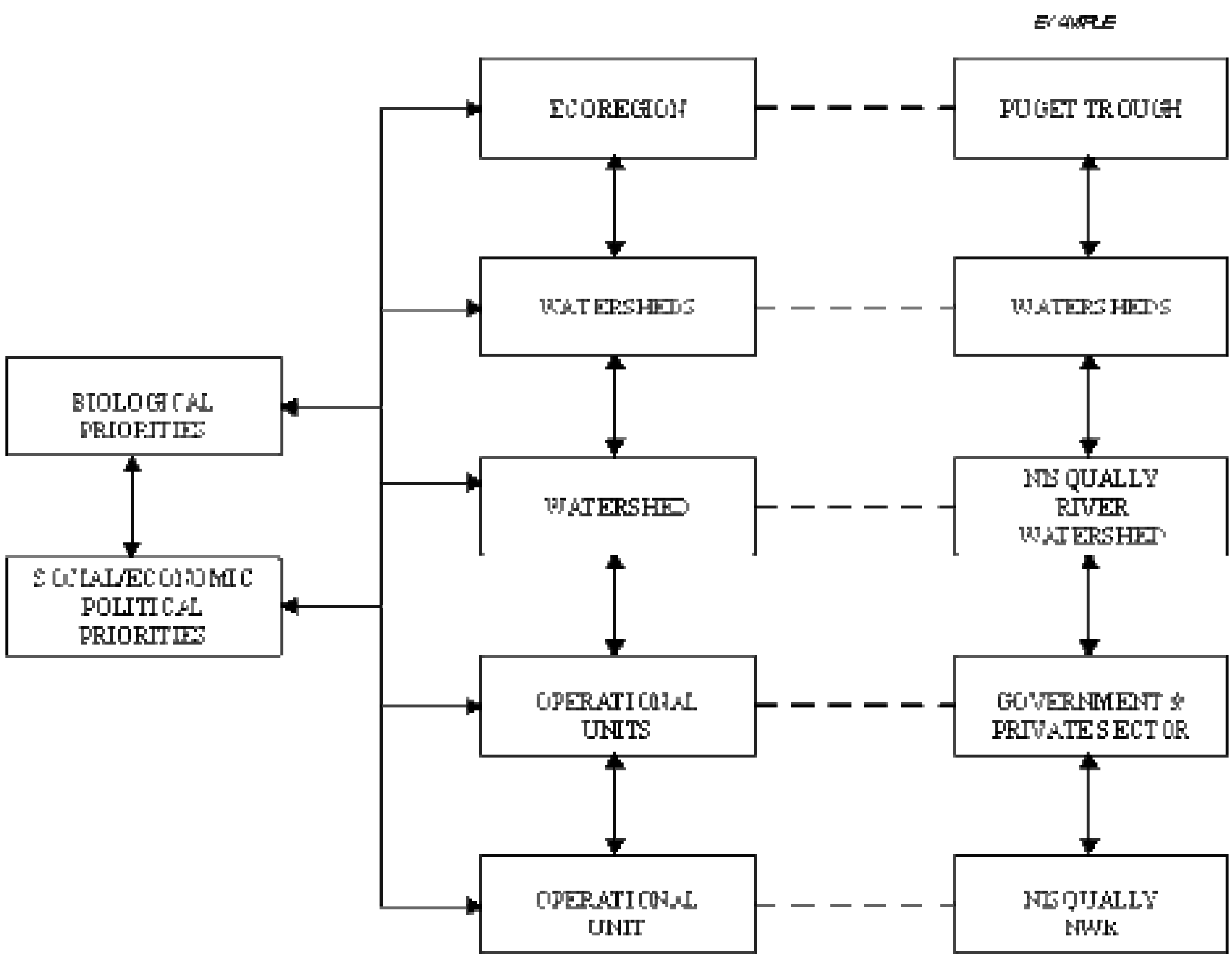
4. The final guidelines for special management area design come not from conservation biology but from the more practical world of park management. The location of special management area boundaries influences essential management activities such as transportation, visitor control, fencing, and controlled burning. Special management area staff, visitors, and researchers all need to move about a special management area without damaging natural communities. Engineering constraints limit the placement and cost of roads and trails. Boundaries should be adjusted to avoid difficult obstacles (canyons, mountains, rivers) between portions of the special management area. Fire burns upslope; when controlled burning is an anticipated management practice, special management area boundaries should follow ridge lines and other natural firebreaks. Many natural areas require fencing to exclude people, livestock, or exotic animals. The cost and ease of fence building is related to topography and soils. Adjusting boundaries to lower the cost of fencing, even if special management area size must be increased, may be cheaper than drilling post holes in lava or granite. Finally, visitor facilities and housing for managers need to be placed on less sensitive parts of nature special management areas. Additional land may be

needed within special management area boundaries for buildings, parking lots, etc.

Natural areas are expected to maintain biodiversity for centuries. The long term expenses of management can easily outweigh the costs of special management area establishment. Making boundary adjustments to minimize management costs is as important to special management area viability as those necessary to maintain population, community, and ecosystem processes.

Conclusions

A clear understanding of the differentiation between identifying a representative natural area network and designing individual viable natural areas will assist development of a national strategy to conserve biodiversity. Regional biodiversity distribution data bases are not intended to convey information about population or ecosystem processes. By definition, these processes are dynamic and can be accurately described only for small areas and short time periods. Special management area designers use detailed information about these local processes to make determinations about the special management area size and shape they hope will endow long term viability on particular natural areas. Recognizing the distinction between biogeographic analyses for natural area network identification and the biological, ecological, and practical analyses that constitute special management area design is the first step toward a consensus for developing a national biodiversity conservation strategy.


EXAMPLE
ECOREGION
PUGET TROUGH
WATERSHEDS
WATERSHEDS
BIOLOGICAL PRIORITIES
WATERSHED
NISQUALLY RIVER WATERSHED
SOCIAL/ECONOMIC POLITICAL PRIORITIES
OPERATIONAL UNITS
GOVERNMENT & PRIVATE SECTOR
OPERATIONAL UNIT
NISQUALLY NWR

Use of Gap Analysis Data to Establish Goals and Priorities for Individual Land Management Units - National Wildlife Refuges in Washington State

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With demands on natural resources increasing, land managers need to adopt a landscape approach in developing management goals and priorities (Fig. 1). Whereas efforts in the past have focused on individual management units in isolation, Gap Analysis data provide a landscape context for land management units, irrespective of land ownership. In this paper, we describe the results of a preliminary analysis of the contributions of three National Wildlife Refuges (NWRs) to the conservation of biodiversity in the ecoregions in which they are located. This project, which will include all of the NWRs in the state when completed, is a cooperative effort between the Washington Gap Analysis Project, the U.S. Fish and Wildlife Service (FWS) Region 1-Refuges and Wildlife, and the FWS's field office in Vancouver, Washington.

(See Figure 1.)

Our preliminary analyses include the Nisqually NWR in the Puget Trough ecoregion on the west side of the Cascade mountains, and the Turnbull and Little Pend Oreille NWRs in the ecoregion referred to as the Northeast Corner (ecoregional boundaries correspond to those described by Bailey [1980] as refined by the USFS and WAGAP). For each ecoregion, we identified the proportion of land in each vegetation zone, the actual land cover within each zone, and the proportion of each zone in each of five conservation status categories. The latter correspond to the National GAP guidelines, except that for this analysis we divided lands not managed for native species into public, e.g., DOD and tribal lands (conservation status 4) and private lands (status 5). We then identified those vertebrate species predicted to occur within the ecoregions and each of the refuges. Vertebrate distributions were based on each species' association with actual land cover. This allowed us to calculate the proportion of each species' predicted distribution on "reserves" (conservation status codes 1 and 2; lands managed for biodiversity) and to develop a "report card" describing the contribution of each NWR to the conservation of vertebrate biodiversity in their respective ecoregions. And finally, based on ecoregional context, we made recommendations as to the management goals and priorities for each NWR, both within and outside their boundaries.

Nisqually NWR

Nisqually NWR, like most of the refuges in the Puget Trough ecoregion, is small and not connected to other areas managed for biodiversity. However, the refuge contains examples of most of the major habitat types within the Puget Trough ecoregion. This habitat diversity accounts for the high proportion of Trough vertebrates predicted to be present (see report card below), but surrounding development threatens to reduce adjacent habitat patches to where they may not support viable populations of some species. Lowland forest (< 2% in reserves) is particularly threatened within the Puget Trough ecoregion, and forested areas on the refuge are in danger of becoming isolated.

(See "Report Card".)

Based on modeled distributions, 45 of the ecoregion's native mammals are predicted to occur on the refuge, including 7 of 9 species listed as threatened or endangered by the state or federal government; 90 of the ecoregion's 144 native breeding birds, including 10 listed species; and 13 of the region's 22 native reptiles and amphibians.

The Nisqually River is the refuge's primary link to larger undeveloped areas. Compared to other large rivers within the Puget Trough ecoregion, the Nisqually has the least surrounding developed and agricultural land. Maintenance of this corridor to other protected areas in the watershed via land acquisition or land-use planning appears to be critical for ensuring the continued contribution of the refuge to the protection of biodiversity in this ecoregion.

Turnbull and Little Pend Oreille NWRs

The conservation status of vegetation zones varies considerably within the Northeast Corner ecoregion (see table below). Statewide, 49 percent of the Ponderosa Pine zone is privately owned. Three percent of this zone is managed for biodiversity in the Northeast Corner ecoregion, compared to 12 percent statewide. The Western Redcedar/Western Hemlock zone also has only 3 percent of its area managed for biodiversity in this ecoregion, but 70 percent of its total area is publically owned. In contrast, 44 percent of the Subalpine Fir zone occurs within "reserves," and only 3 percent of its total area in this ecoregion is privately owned.

(See "Conservation Status".)

Turnbull NWR is almost entirely within the Ponderosa Pine zone. One of its major assets is its status as one of the few conservation areas with this forest type. The refuge is, however, on a "peninsula" of Ponderosa Pine forest among agricultural lands and steppe, and development around Spokane threatens to isolate the refuge from other forests. Fifteen of the ecoregion's 16 reptiles and amphibians are predicted to occur on Turnbull NWR, as are 46 of 64 native mammals, and 105 of 160 species of breeding birds (see report card). Ten listed species of mammals and birds are predicted to occur on the refuge. Management recommendations from this preliminary analysis include maintaining existing grasslands and open canopy Ponderosa Pine woodland on the refuge and, if possible, preventing isolation from other forests to the north.

Little Pend Oreille NWR contains all of the major forest zones and forested habitats within the ecoregion. Not only is it the largest refuge in the state, it is bordered by national forest to the north and south. Because of its size and location, it has greater potential than smaller refuges to support large animals or those with large home ranges. Probably the refuge's greatest deficiency is its lack of connection to habitats along the Colville or Little Pend Oreille Rivers. Most of the reptile, amphibian, and mammal species in the ecoregion and 94 species of breeding birds are predicted to occur in the Little Pend Oreille NWR. Our preliminary analysis indicates that maintenance of a corridor to adjacent river valleys would help maximize the contribution of the refuge to biodiversity protection.

Overall, the three refuges are predicted to provide some habitat for 38 percent of the state's listed species and 80 percent of the remainder. We note that predicted presence does not necessarily mean that the species are confirmed as present or that the habitat on the refuge has been confirmed as suitable. More detailed field-level sampling is needed for the next stage of conservation planning. This analysis is an example of how to begin the planning at the ecoregion and landscape levels.

We believe our analysis, when completed, will serve as a model for the application of GAP data to the development of management goals and priorities within the National Wildlife Refuge System. Similar analyses for Fort Lewis and Camp Bonneville (both belonging to the U.S. Department of Defense) have been well received. The latter was recently considered for addition to the National Refuge System.

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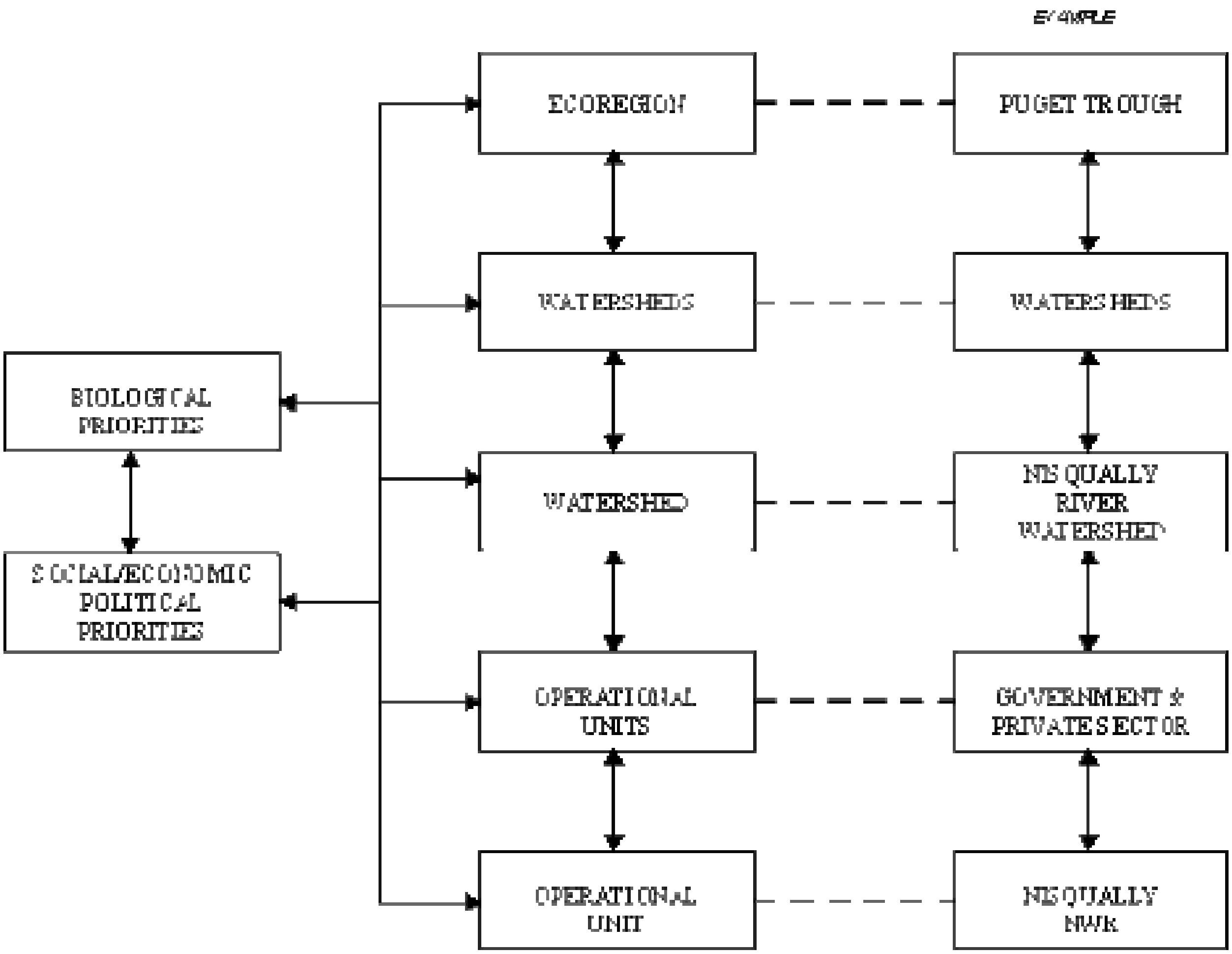

FIGURE 1

REPORT CARD FOR NISQUALLY, TURNBULL, AND LITTLE PEND OREILLE NWRs

	Herps		Birds		Mammals	
	Listed*	Other	Listed	Other	Listed	Other
Puget Trough	6	16	21	123	9	41
Nisqually NWR	0	13	10	80	7	38
Northeast Corner	3	13	25	135	15	49
Turnbull NWR	3	12	10	95	10	36
Little Pend Oreille	3	10	11	83	10	43
State	21	24	55	172	31	70
3 NWRs	3	18	19	137	18	59

* Listed Includes federal and state listed species

CONSERVATION STATUS IN WASHINGTON STATE FOR ZONES OCCURING IN TURNBULL AND LITTLE PEND OREILLE NWRs

	1	2	3	4	5
PIPO	2	1	25	23	49
PSME & AMGR	6	2	45	13	34
THPL & TSHE	2	1	67	0	30
ABLA & ALPINE	43	1	41	12	3
STATEWIDE	11	1	25	6	57

Numbers are percents.
PIPO = Ponderosa Pine, PSME & ABGR = Douglas-fir/Grand Fir, THPL & TSHE = Western Redcedar/Western Hemlock, ABLA = Subalpine Fir.

Modeling Grizzly Bear Habitat Suitability in Idaho

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Many of the issues confronting wildlife managers and scientists are challenging the conventional spatial boundaries defined by administrative units. This holds especially true in the management of large carnivores such as wolverines, wolves, mountain lions, and grizzly bears. Individual grizzly bears range over 400 to 1000 square kilometers in a lifetime, while viable bear populations may require 10 to 30 times as much space. Such scales require a very broad view of habitat conditions. Not insignificantly, understanding these bears requires regional GIS databases that transcend state and even national boundaries.

Idaho is currently grappling with a number of issues related to grizzly bear management, including the potential reintroduction of a population into its central mountain wilderness areas and the management of humans in areas currently occupied by grizzly bears in the Panhandle and in the Yellowstone ecosystem. There has been reoccurring debate over the extent and location of "suitable" habitat. In addition, there are concerns about fragmentation and insufficient overlap between physically productive habitat and wilderness areas secure from substantial human intrusion. Scientists from the University of Idaho's College of Forestry, Wildlife, and Range Sciences GIS Lab and from the National Biological Service's Cooperative Park Studies Unit are trying to answer to these questions and develop a prototype for looking at the suitability of habitat for large carnivores elsewhere.

This research has drawn upon regional GIS databases, including GAP data for the state of Idaho, to model grizzly bear habitat suitability. These data were rasterized and combined in ARC/INFO grid format. Because grizzlies, like most other large carnivores, die primarily because humans kill them, a large part of this model deals with human-related features such as townsites, roads, and trails. This information is integrated into a measure of potential human activity for each map pixel and treated as an analogue of grizzly bear death rate. Information on vegetation, topography, and ungulate populations is integrated into seasonal measures of potential habitat productivity and treated as an analogue of birth rate. These two metrics are then combined in a way that culminates the analogy—by subtracting the standardized index of human activity from the standardized index of habitat productivity, the resulting measure is a direct analogue to population dynamics.

This model has already produced information of value to management deliberations. Maps have been produced that show seasonal habitat productivity for the entire state, as well as the location of "suitable" habitat defined by increasingly restrictive criteria. These maps show that, by most standards, there is abundant well-protected grizzly bear habitat in central Idaho that could potentially support a reintroduced bear population. They have also highlighted the potentially precarious status of existing grizzly bear populations, especially in the Panhandle. These results, as well as a description of the method, are parts of a manuscript that is currently being reviewed prior to submission to a journal for publication.

Even though significant progress has been made with this project, some major work remains ahead. In particular, we are prioritizing efforts to relate model outputs to parameters more directly relevant to management considerations, including actual grizzly bear birth and death rates. To date, we have partially confirmed the model by comparing outputs with delineations of currently occupied habitat and by assessing statistical relationships with bear sightings. We anticipate substantial future progress by extending the method to well-studied bear populations in areas such as the Yellowstone ecosystem and the northern Rocky Mountains of Montana.

Gap Analysis of the Vegetation of the Intermountain Semi-Desert Ecoregion

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The nation's first formal Gap Analysis of a multistate ecoregion has been conducted for the Intermountain Semi-Desert ecoregion (Bailey 1995). The Intermountain Semi-Desert ecoregion encompasses approximately 412,000 km^2 in portions of Washington, Oregon, Idaho, Nevada, California, Utah, Wyoming, Colorado, and Montana. Two geographically disjunct subregions make up the ecoregion, the Columbia Plateau in the west, and the Wyoming Basin in the east. The Intermountain Semi-Desert boundary corresponds closely to the limits of Küchler's sagebrush steppe potential natural vegetation type. The southern boundary of the Intermountain Semi-Desert ecoregion grades into the Intermountain Semi-Desert and Desert Province, which tends to be warmer, drier, and with greater topographic relief than the Intermountain Semi-Desert ecoregion. The Cascade and Sierra Nevada mountain ranges bound the ecoregion on the west and the northern Rocky Mountains bound it on the north and east.

This ecoregion was selected for the prototype regional gap analysis for both practical and conservation reasons. From a practical standpoint, the Intermountain Semi-Desert ecoregion was among the first for which the requisite land cover and land management mapping were completed by the individual state-level GAP projects. Additionally, the area provides a suitable testing ground for demonstrating whether GAP can overcome the technical challenges associated with large-area regional mapping that have concerned some program reviewers. Very little land in the Intermountain Semi-Desert ecoregion has been designated for maintenance of biodiversity, while potentially conflicting land uses such as grazing and cultivation are extensive. Enough undeveloped habitat remains, however, for pro-active conservation action to be effective. Thus the Intermountain Semi-Desert ecoregion makes a representative case study that could be applied to other regions throughout the western U. S. Planning for conservation and ecosystem management within this ecoregion is under way by The Nature Conservancy, the Oregon Biodiversity Project, and the Interior Columbia Basin Ecosystem Management Project (a joint effort by the U. S. Forest Service and Bureau of Land Management). Also, BLM is considering wilderness proposals in Wyoming, and proposals for other new wilderness areas and national parks in Idaho and Wyoming are being discussed. A regional Gap Analysis could add valuable information for all of these planning programs.

Land cover was originally mapped independently for each of the states in the Intermountain Semi-Desert ecoregion. Although most state GAP projects used 1990 (+/- 2 yrs) satellite imagery from the Landsat Thematic Mapper sensor, combined with field inventories and existing maps of vegetation in compiling their land cover data, they differed in methods and products. Maps for Idaho and Oregon used photointerpretation techniques with older, lower-resolution Multispectral Scanner images and had larger minimum mapping units than the other states. In contrast, land cover mapping in Nevada and Utah was done with digital image processing of TM image mosaics. This digital classification approach generally achieved greater spatial resolution at some expense in classification detail. The other state projects fall somewhere in between these methods, using manual photointerpretation of higher resolution TM data.

Experienced GAP staffers from states across the ecoregion collaborated to compile and standardize the database and to conduct the analyses. A workshop was held at the University of California, Santa Barbara (UCSB), in June 1996 for the members of this ecoregional team, led by Frank Davis and David Stoms at UCSB. The group first cross-walked the state land cover types to a standardized set of alliances—or to a higher level of classification when necessary. A preliminary regional map was generated by mosaicking the cross-walked state maps together. Then the CA-GAP staff developed an innovative mapping technique to produce a regional land cover map with greater spatial and thematic consistency. Multitemporal satellite imagery from the NOAA Advanced Very High Resolution Radiometer (AVHRR) was used to refine the preliminary map by providing a more consistent spatial resolution (1 km^2 or 100 ha pixel size) across the entire Intermountain Semi-Desert ecoregion while retaining its basic floristic information. This mapping technique is described in Stoms et al. (in review). The team also assisted in compiling a consistent regional land management status map, which required some standardizing of definitions and attributes.

The total amount of land permanently protected in the ecoregion is less than 4%, and most types characteristic of the region have less than 10% of their area represented in conservation lands. Of 48 land cover types, twenty were found to be particularly vulnerable to potential loss or degradation because of the low level of representation in biodiversity management areas and the likely impact of land use activities. The gap analysis data and findings (described in Stoms et al. in press) will be useful in providing a regional perspective in project impact assessment and future conservation planning within this ecoregion.

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Gap Analysis for Ant Species

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Introduction

Clearly, when we speak of biodiversity, it is appropriate to consider plant and invertebrate diversity as well as vertebrate diversity. Vertebrates account for less than 2% of presently described animal species (Gaston 1991). Almost all undescribed species are invertebrates. Few of the estimated 27,000 species going extinct each year are vertebrates (Wilson 1992). Vertebrates utilize relatively large home ranges likely to span several vegetation and habitat types. Most vertebrates, even habitat specialists, are habitat generalists when compared to invertebrates. The scale of perception and environmental exploitation of invertebrates is orders of magnitude smaller than that of vertebrates. Furthermore, the ability of technicians to classify vegetation types exceeds the resolution of habitat utilization by vertebrate species (Maser et al. 1984). For example, the Florida Biological Diversity Project is using a habitat classification scheme that recognizes >100 plant associations. At this level, few, if any, vertebrates are specific to any one association (C.R. Allen, unpublished data), and most species span numerous associations.

Nodes of high biological diversity determined from vertebrate species richness are likely to be in the range of 100s to 1000s of hectares (e.g., Cox et al. 1994). Decisions concerning land use, habitat protection, and conservation are likely to be an order of magnitude smaller. Additionally, small areas unable to support a large variety of terrestrial vertebrates may nonetheless be species-rich (containing a high richness of plant and invertebrate species). Land-use and conservation decisions made using vertebrates as indicators of biodiversity will realistically assess impacts on or protect vertebrates but may have little usefulness in conserving overall biodiversity.

The case for using arthropods for the inventory of biodiversity has been convincingly made (Kremen et al. 1993). Prendergast et al. (1993), in an examination of species richness in Great Britain, compared the diversity hot spots of birds, mammals, butterflies, and liverworts and found that the species-rich areas within each taxon rarely overlapped. Landres et al. (1988) cautioned against the use of vertebrates as an index of biodiversity; a range of well-chosen organisms that will explicitly better represent overall biological diversity is needed in an index of biological diversity. Due to the vast number of described invertebrates, it would be impossible to include them all in initial efforts. Therefore, invertebrate groups should be carefully chosen to maximize their contribution to determining overall patterns of biodiversity.

Butterflies have been suggested as an invertebrate taxonomic group to utilize for biodiversity monitoring (Kremen 1992). However, data from Florida indicate that despite the host-plant specificity of some larval forms, adult butterflies are mostly edge-type species with little overall habitat specificity. Indeed, birds may be more habitat specific than butterflies (Debinski and Brussard 1994). Among the invertebrates, the Formicidae exhibit many traits that make them an excellent choice for inclusion in an index of biodiversity. Ants are relatively easy to identify, and a relatively short period of training and adeptness with a dichotomous key will enable most persons to identify the ants of temperate regions. In Florida there are approximately 190 ant species, similar to the number of herpetofauna and avian species. Ant species are easy and inexpensive to sample, and a variety of established sampling methodologies exist. Additionally, the range and habitat affinities of ants are well known when compared to families such as Scaraebidae (an estimated 250 species in Florida; Woodruff 1973), or Staphylinidae (an estimated 450 species in Florida; Frank 1986).

Ants act as keystone species in many instances (Risch and Carroll 1982). They provide key and irreplaceable ecosystem services such as pollination, nutrient turnover, energy flow, and seed dispersal (Handel et al. 1981). The Formicidae exhibit a wide range of habitat specificities and diversity of lifestyles. Some species utilize very specialized microhabitats (see below), and feeding niches are likely to be saturated (Holldobler and Wilson 1990). Because of niche saturation, the Formicidae are excellent indicators of fine-scale habitat heterogeneity, which in turn is an excellent indicator of biological diversity. Additionally, niche specialization means that general ant sampling may be used to bioassay ecological trends by monitoring trends of species with specific life-history traits of interest. Ants, indicative of terrestrial habitat heterogeneity, and birds, indicative of structural heterogeneity across habitats (e.g., Cyr and Cyr 1979), may make an excellent pair of organisms for indexing biological diversity. Short generation time in ant species translates to rapid response to environmental change. These positive characteristics are good for inventory and monitoring and allow for a finer-scale resolution in biodiversity mapping. The inclusion of the Formicidae in an index of biodiversity yields a broader base and more precise information for cross-scale decision making.

Several species present in Florida illustrate the potential of the Formicidae as indicators of biodiversity. The tropical fire ant, *Solenopsis geminata*, acts as a keystone species (Risch and Carroll 1982) affecting invertebrate community composition where it is not displaced by the exotic *S. invicta*, which itself acts as a keystone (Wojcik 1994). Several species of ants are endemic to Florida, and others are nearly so. *Paratrechina wocjiki* is endemic to central Florida where it may be found in a variety of habitats (Deyrup and Trager 1986). *Conomyrma flavopectus* is restricted to sugar sands in central Florida with early successional stages of sand pine (Trager 1988). *Paratrechina phantasma* is endemic to Florida scrub and dune habitats (Trager

1984). Four species of ants are endemic to the unique Florida scrub and sandhills habitats; this exceeds the number of endemic scrub species for any of the vertebrate taxa or for butterflies.

Many species are habitat-specific. *Xenomyrmex floridanus* is restricted to mangrove (Deyrup et al. 1989). *Leptothorax allardycei* is limited to sawgrass and *Crematogaster vermiculata* to cypress (Deyrup et al. 1989); no vertebrates are sawgrass or cypress specialists. *Leptogenys manni* is endemic to Florida and feeds only on isopods (Trager and Johnson 1988); such niche specialization is not unusual, given the general niche saturation found in the Formicidae, and may be useful in ecological monitoring.

The predaceous species *Odontomachus clarus* illustrates interesting biogeographical patterns (mirrored by some much studied and heralded vertebrates, such as the Florida Scrub Jay). *Odontomachus clarus* is known only from xeric upland areas of Mexico, the southwestern United States, and subtropical Florida (Deyrup et al. 1985). Florida Formicidae are largely a mix of temperate continental species and species of West Indian origin, a pattern also seen in Florida's birds and butterflies.

Because vertebrates and invertebrates interact with their environment at different scales, there is a critical need to include some invertebrate taxa in an index of biodiversity, and ants are a desirable and defensible taxa to use. Recently the USGS-BRD has considered mapping the Formicidae at a national level. Here we present two different methodologies for spatial mapping of ant diversity and a comparison of patterns of species richness between ants and mammals in southern Florida.

Methods

Literature-based (Florida): Geographic distribution of species (i.e., ants and mammals) was determined at the county level. For ants, distribution was determined primarily from published sources (Buren and Whitcomb 1977, Carroll 1975, Cole 1982, Creighton 1950, Deyrup 1991, Deyrup and Trager 1986, Deyrup et al. 1988, 1989, Johnson 1986, Klotz et al. 1995, MacKay 1993, Samways 1983, Schneirla 1944, Smith 1930, 1933, 1944, 1979, Thompson 1989, Thompson and Johnson 1989, Van Pelt 1947, 1950, 1956, 1958, 1966, Watkins 1985, Wheeler 1932, Wilson 1964) and from the unpublished data of D. P. Wojcik and C. R. Allen. The availability of data varied by county. For several counties largely in private ownership with limited access, little data was available, and for some other species distribution is poorly known. We interpolated distributions in counties lacking data based on the presence or absence of species in adjacent counties or known biogeographic affinities of species. These data were then used to produce a county x ant species matrix. All resulting county-level distribution maps were reviewed by recognized experts.

Habitat affinities for ants also were determined primarily from literature review. The Florida bibliography of species habitat use and ecology includes >1300 sources (too many to cite, but the bibliography may be accessed at http://coop.wec.ufl.edu/gap) which have been used to create descriptors of habitat use by species, including ant species. These data were then used to produce an ant species x land cover type matrix. In conjunction the two matrices were then used to produce habitat-specific spatial distributions of all ant species present in Florida, as well as a map of overall ant species richness.

Sample-based (South Carolina): In South Carolina the most recent (and only) comprehensive documentation of ant distribution appeared in 1916 (Smith 1916, Smith and Morrison 1916). This general lack of data in South Carolina necessitated that South Carolina take a sample-based approach to mapping ant diversity, which is currently under way. Ants are sampled throughout the state of South Carolina. Sampling is stratified by physiographic region (sandhills, coastal plain, piedmont, mountains) and by generalized South Carolina Gap Analysis land cover types (n = 28). Ten replicates (randomized within the constraints of access to some properties) in each land cover type in each region will be sampled for a total of approximately 1120 sampled habitat patches across the state (not all land cover types are represented in each strata and some may be of minor importance). Each habitat patch will be sampled by establishing a linear transect consisting of multiple sample points. Sample points will consist of bait samples and pitfall traps. Together, these two sampling methods will capture a majority of ant species present in the state. Pitfall sampling is the better method of sampling overall ant diversity, as aggressive species (e.g., red imported fire ants) will preclude other species from baits. We will also conduct limited sampling with other methods, such as arboreal (C.R. Allen, unpublished manuscript) and subterranean sampling. At each sample point, data on habitat also will be collected. Results of these sampling efforts will be used to simultaneously determine both the county-level distribution and habitat affinity of each species.

Ant sampling is relatively easy and fast. However, identification of species can be problematic. To successfully produce a sample-based data set of ant distributions for a GAP layer, we have had to establish a highly cooperative effort. In this case, cooperators include Clemson University's Department of Aquaculture, Fisheries and Wildlife and Department of Entomology, the South Carolina Cooperative Fish and Wildlife Research Unit, the South Carolina Gap Analysis Program, the National Gap Analysis Program, the USDA Agricultural Research Service, and the South Carolina Department of Natural Resources.

Spatial correspondence between ants and mammals in south Florida

One example of how the data are used is in comparing ant distributions with distributions of mammals. Land cover for the lower peninsula of Florida was mapped at 30-m resolution from classification of 1993 and 1994 Landsat Thematic Mapper satellite imagery. Bands 2,3,4, and 5 of the imagery and a tasseled cap transformation were used in an iterative unsupervised clustering algorithm. Labeling of the spectral clusters with vegetation associations followed the National Vegetation Classification (Grossman et al. 1998, FGDC 1997) to the alliance level (Weakley 1997). Labeling was assisted by auxiliary information such as land use/land cover maps from the South Florida Water Management District, National Wetlands Inventory maps, soils maps, and vegetation surveys and photo-interpreted points from low altitude aerial videography in Everglades National Park and Big Cypress National Preserve (Figure 1).

Note: Figures can be viewed at http://coop.wec.ufl.edu/GAP/antmammal_spatial_corr.htm.

Ants and mammals were modeled in similar ways, following Gap Analysis procedures (Scott et al. 1993) and as outlined for ants above. We produced species richness maps of both taxa (Figures 2 and 3). Richness of both taxa was normalized such that the highest richness for each taxon was equal to one and the lowest richness equal to zero, so that the two coverages were comparable. A coverage of spatial correspondence was then produced by subtracting the normalized mammal species richness map from the normalized ant richness map.

Results: In the coverage of spatial correspondence (Figure 4), values near 0 (green) reveal that richness between mammals and ants are equivalent. High positive values (red to orange) identify areas with higher mammal richness relative to ant richness, and high negative values (blue to magenta) identify higher ant species richness relative to mammal richness.

Comparisons of mammal and ant species richness reveal interesting patterns of correspondence and disharmony between the two taxa. The large areas of green on the correspondence coverage indicates that richness between mammals and ants was similar over much of the Florida Everglades (but see below). However, two interesting deviations occur. In the Big Cypress area of southwest Florida, there is a lack of correspondence between mammals and ants, primarily in cypress-dominated habitats. This is not necessarily because mammal species richness is especially high in these areas, but because ant richness is low. Further north, the opposite situation exists; normalized ant species richness is higher than normalized mammal species richness in several pine-dominated habitats. In most terrestrial habitats (excluding the saturated everglades habitats which constitute a large area of south Florida), spatial correspondence between ants and mammals is low.

Discussion

Invertebrates contribute far more to overall species richness than do vertebrates, and nodes of high richness among different taxa are likely not to correspond. This mandates the inclusion of invertebrates in an index of biodiversity. Among the Arthropoda, the Formicidae are a good family of choice for mapping because data are available or relatively easy to obtain, they utilize a wide variety and large number of niches, and some ant species are very habitat- and condition-specific. Utilizing the Formicidae in biodiversity mapping efforts offers the chance to increase the thematic resolution when representing geographic nodes of high species richness. Future land-use decisions will likely be at a scale an order of magnitude smaller than decisions made in the past. The inclusion of the Formicidae in programs investigating biodiversity assures that land-use decisions will be made utilizing species information applicable across a range of scales.

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Gap Analysis in Riverine Environments

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Background

As Jennings (1997) mentioned in Bulletin No. 6, the National Gap Analysis Program is in the initial stages of developing the aquatic component of Gap Analysis. This effort, to date, has included drafting general technical documents to guide the development of pilot projects as well as establishing two pilot projects. These include a project for the upper Allegheny River basin in western New York, initiated in 1995, and a statewide project for Missouri, initiated in 1997. The purpose of this article is to outline the basic approach developed by the Missouri Resource Assessment Partnership (MoRAP).

The project addresses several objectives; however, the three primary ones are: 1) develop an objective method for identifying gaps in biodiversity conservation in riverine environments, and set priorities for filling those gaps, 2) identify problems of and methods for effectively integrating the terrestrial and aquatic components of Gap Analysis, and 3) document information needs, successes, failures, obstacles, time, and costs, which will assist other states with similar efforts.

We established some priorities for our project to make it more reasonable in scope and to help us maintain a more structured approach. First, we are strictly focusing on riverine environments. Missouri is essentially a "stream state" with the majority of our aquatic biodiversity concerns situated in riverine environments. Second, although Gap Analysis will continue to include all taxa, this project focuses on fish, mussels, crayfish, and snails. These four taxonomic groups were selected primarily because of the availability and quality of existing sampling data.

Approach

There are five major steps to the approach we developed. The first step involves delineating and mapping a 1:100,000 digital data layer of valley segment types for Missouri. These valley segment types can be viewed as the lotic counterparts of wetland or lake type classifications (Figure 1). To accomplish this step, we are using the Aquatic Community Classification System developed by The Nature Conservancy (Lammert et al. 1996). This hierarchical classification system focuses on ecological regions and hydrogeomorphic variables to delineate distinct valley segment types. Using this digital data layer, we will then generate the critical base line inventory statistics required for conducting accurate assessments and developing meaningful conservation priorities. These statistics include how many valley segment types there are within each ecological section (Bailey 1980), how many miles there are of each type, and where they are (Figure 2).

The second step incorporates digital instream and watershed management and land use information into a coarse-level analysis process (Figure 3). Essentially, the question being addressed in this step is how well are we currently conserving each of Missouri's valley segment types. Answering this question requires relativistic comparisons using percentage statistics: for example, calculating the percentage of each valley segment type currently in the public trust, the percentage of the total stream miles each valley segment type represents within each region, and the percentage of each valley segment type that can be classified as high-quality. There are numerous other calculations like these that will be incorporated into this step of the process. The key point is that this step needs to be flexible enough to meet a wide variety of user needs yet remain focused on establishing first-cut biodiversity conservation priorities for each region.

As can be seen in steps one and two, we are not using biological information to develop our initial conservation priorities. Our initial reasoning is that all types of the various riverine environments must be considered as having conservation value, regardless of their biological communities. Therefore, we first focus on the rarity and the conservation status of riverine types themselves. Consequently, riverine environments inherently low in diversity are weighted equally with those inherently high in diversity at the outset. This avoids the problem of developing conservation priorities which simply focus conservation efforts on those inherently diverse environments while ignoring very unique environments that may have relatively simple communities. I believe that the number of rare species tends to be strongly associated with community diversity, at least in aquatic environments; however, this relationship is probably scale-dependent. Even conservation assessments that focus on rare, threatened, or endangered species will tend to establish conservation priorities favoring inherently diverse environments rather than unique environments. This may not be true for all regions; for instance, the very simple communities of the Arid Southwest harbor a large number of rare aquatic species. However, this association between rarity and diversity definitely holds true for the majority of the nation as well as those regions covering Missouri (Figure 2).

Once conservation gaps are identified for the valley segment types and initial priorities established, we move on to step three in which distributional data for all known species of fish, mussels, crayfish, and snails are used to predict the community potential of each individual valley segment in the state. To accomplish this difficult task, we need three different pieces of information in digital format (Figure 4). First, we need to know the statewide distribution of each species. More specifically, we need to know all of the watersheds (14-digit Hydrologic Units; USDA 1992) in the state in which a given species exists. Second, we need to know all of the general habitat requirements or affinities of each species so we can predict their distribution throughout the watersheds in which they are known to occur. (Few species are found throughout entire watersheds; most reside in specific segments of watersheds such as headwater segments or only warm water segments.) Finally, we need the valley segment data layer, which provides the habitat type template for predicting local distributions. This is analogous to the land cover layer for predicting the distribution of terrestrial species. The end product of this exercise will be a 1:100,000 digital data layer of valley segment types for Missouri, with each segment attributed with the fish, mussel, crayfish, and snail species likely to occur in that segment under pristine conditions (Figure 5).

In step four we use the distributional data developed in step three to revise our initial conservation priorities (Figure 6). Specifically, we use this information to identify specific locations of “high-priority” valley segment types which: a) are relatively high in species richness, b) serve as centers of endemism, or c) harbor species of special concern. For simplicity’s sake, we label these valley segments as “segments of biological significance.” Like the second step, step four must also remain flexible to meet a wide variety of user needs. This fourth step is necessary and important because the same valley segment type will be found in many different locations and, due to zoogeographic factors, these different locations will often have different biological assemblages. This fourth step thus serves an important role in further refining management options since even slight differences in species assemblages among locations may have a significant effect on decisions related to biodiversity conservation.

The final step in the overall process involves further refining our conservation priorities by identifying specific valley segments which are both biologically significant and high-quality examples of a particular valley segment type (Figure 7). The resulting final maps will then show the locations at which our biodiversity conservation efforts should be focused and where we can assume we will most likely succeed. To assess the relative quality of each valley segment, we will develop “quality-ranking” models. To accomplish this we are working with resource professionals from around the state to identify the major "stressors" and management activities within each ecological section that either positively or negatively influence our riverine environments. Once identified, we will compile digital data for those major stressors and management activities. We will then develop a protocol, to account for these major stressors and management activities, which examines both the local conditions surrounding a given segment (e.g., is it channelized or lacking a natural riparian corridor?) as well as the condition of its surrounding watershed (e.g., percent forested, road density, potential pollution sources, acres of CRP, etc.).

When this five-step process is completed, thousands of miles of stream will have been examined, resulting in a workable number of high-priority stream segments on which to focus conservation efforts. The three most important aspects of the aquatic component of Gap Analysis are: 1) it provides an objective (i.e., data-driven) approach for assessing biodiversity conservation needs, 2) it provides the common framework necessary to make truly relative conservation assessments across states, regions, watersheds, etc., and 3) it has built-in flexibility to account for a wide variety of user needs. Information from the Missouri project will assist state and federal resource agencies in making decisions pertaining to new land acquisitions, new management plans, or for identifying focus areas for land owner incentive programs and research.

For more information, see http://www.cerc.cr.usgs.gov/morap/ or con-

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Final Report Summary: New Mexico Gap Analysis Project

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This research included all of New Mexico, a 314,920 km^2 landscape that reflects a varied geologic and natural history. New Mexico's diverse array of species is attributable to complex connections of regional biogeographic components from the Great Plains, Rocky Mountains, Great Basin, and Chihuahuan and Sonoran Deserts.

Land Cover Classification and Mapping

We developed our land cover classification scheme in cooperation with the New Mexico Natural Heritage Program and consultation with experts on New Mexico vegetation. The final land cover map has approximately 24,260 polygons representing 42 mapped classes that include 33 terrestrial and riparian vegetation community classes, 2 hydrologic feature classes, 2 aquatic classes, 2 urban classes, and 3 classes of barren, rocky, or mined ground. We assessed accuracy of the final land cover map during February-July 1995 by ground-truthing 1,763 polygons with cooperation from state and federal agencies and a variety of other knowledgeable and interested people. Conservative accuracies among mapped classes ranged from zero to 80% at grouped cover-type level. Highest accuracy was associated with agricultural land cover, high-elevation conifer forest, urban vegetation, desert scrub, and natural surface waters. Accuracy among classes generally improved dramatically by accounting for ecotones and inclusions.

Predicted Animal Distributions and Species Richness

We modeled 584 species (26 amphibians, 96 reptiles, 324 birds, and 138 mammals) relative to species-specific data on associations with land cover types, mountain ranges, watersheds, elevation, slope, water, soils, and known general range. We consulted experts to review first-draft maps of species distribution predictions. To assess distribution predictions, we obtained species occurrence data for birds in a county in the northwest corner of New Mexico and amphibians, reptiles, and birds of a military reservation in southern New Mexico. Comparison of predicted animal presence to records of occurrence ranged from 53.8% to 88.6% accuracy among three taxonomic groups for two locations. Omission errors were more prevalent for the county data, whereas commission errors were more prevalent for all taxonomic groups compared for the military area. These patterns related to degree of recent specific surveys of test areas.

Considering all 584 animal species included in our project, we predicted the richest areas in the state to contain 327 species, 56% of the total. Richest areas among taxonomic groups contained 53.8% of 26 amphibian species, 59.4% of 96 reptile species, 65.7% of 324 bird species, and 47.8% of 138 mammal species. Assessment of data for breeding distribution of birds relative to year-round distribution of birds indicated distinctions between those data sets for drawing conclusions about bird richness.

Land Stewardship and Management Status

We used a public domain map of land ownership categories in New Mexico at 1/4-1/4 section (40 acre or 16 ha) resolution. With additional data about specific stewardship boundaries incorporated from federal and state agencies, several tribes, land trusts, and private landholders, we added 670 polygons to the ownership. Before assigning management status categories, we assessed views about management classification from a work group of various federal and state government agencies, tribal representatives, environmental organizations, and private landholders statewide. From variability in the responses, we concluded that the land management categories are not interpreted and applied in the same way by all individuals. Thus, we developed a dichotomous key to consistently assign status to the stewardship boundaries (Crist et al. 1996).

Private lands (45%) were the dominant category of stewardship; federal stewardship was dominated by Bureau of Land Management, U.S. Forest Service, and military lands. We identified 18 general categories of land tracts represented in management status 1 and 2 lands. These categories included an array of federal, state, and private managing entities. Distribution of management status in New Mexico was estimated as 2,418 km^2 of Status 1 (1%), 19,354 km^2 of Status 2 (6%), 89,833 km^2 (29%) of Status 3, and 203,320 km^2 (65%) of Status 4.

Analysis Based on Stewardship and Management

Management status 1 and 2 represent about 7% of the New Mexico landscape. We identified 11 natural land cover classes each represented by less than an estimated 100,000 hectares. Six of these restricted classes (Madrean Lower Montane Conifer Forest, Madrean Closed Conifer Woodland, Broadleaf Evergreen Interior Chaparral, Graminoid Wetlands, Riverine/Lacustrine, and Basin/Playa) each had less than 10% of their estimated area in Status 1 and 2. Statewide, 20 natural land cover classes each had less than 10% of area in Status 1 and 2. Of these classes, nine (primarily Madrean Forest and Woodland, Interior Chaparral, Broadleaf Sand-Scrub, and various Wetlands) each had less than 10,000 hectares in Status 1 and 2 areas. Management Status 1 and 2 lands were nearly all distributed among a variety of federal agencies and functions. Private and tribal stewardship is significant in the overall distribution of many land cover classes; 5 of the 11 most restricted classes have at least 45% of area on private and tribal lands.

We identified 35 species with no more than 1% of their predicted distribution on Status 1 and 2 lands. Nearly 45% of these species were reptiles and amphibians, despite those taxonomic groups representing 21% of all species included in analyses. Six of the nine species with no predicted distribution on Status 1 and 2 lands were amphibians and reptiles which have restricted distributions in southern New Mexico. Overall, 465 species (79.6%) each had less than 10% of their distribution on Status 1 and 2 lands. Importantly, all users of these data should recognize that some species primarily distributed on Status 3 and 4 lands adequately meet their biological needs there. Judicious evaluation will be needed to determine which species represent biological gaps.

Data Use and Availability

NM-GAP data are presented in a format that will operate on a PC configured to run ARC/INFO and ArcView current to November 1996. However, all possible combinations of data queries were not tested. A workstation may be necessary for some operations. NM-GAP data products and documentation may be acquired from the Resource Geographic Information System (RGIS) of New Mexico at (505)277-3622; Internet at http://rgis.unm.edu:8080, or from the national GAP Home Page on the Internet at http://www.gap.uidaho.edu/gap.

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Final Report Summary: Washington Gap Analysis Project

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We conducted the Washington State Gap Analysis within the context of 31 vegetation zones: 9 steppe, 9 westside mesic-wet forest, 11 eastside dry-mesic forest, and 2 high-elevation zones. Data and results are reported in both hard copy and digital format. The hard-copy format is a five-volume report (in press). Volume 1 is a description of current land cover and its conservation status. Volumes 2, 3, and 4 are atlases for herpetofauna, mammals, and birds, respectively, and Volume 5 is the gap analysis. Digital data will be available through the Washington Department of Fish and Wildlife.

Land Cover

Actual land cover within each vegetation zone was mapped by on-screen digitization using spectrally clustered 1991 Landsat satellite Thematic Mapper (TM) imagery as a backdrop. The protection status of each zone was assessed using: 1) the percent of the zone in Conservation Status 1 and 2 lands, and 2) a Conservation Priority Index: CPI = ((100 - % protected)/(100 - % converted)) * log(total area in the zone) where "% converted" refers to the percentage of the zone converted to agriculture or development and "% protected" refers to the percentage of the zone in Status 1 and 2 lands.

Statewide, the percentage of lands in Conservation Status 1 and 2 is 12%, but protected lands are unevenly distributed among vegetation zones. The six steppe zones (all < 6%, four < 1%) and the four Puget-Willamette Trough zones (all < 3%) have the least Status 1 and 2 lands. The percentage of Status 1 and 2 lands in other zones generally increases with elevation, with the Permanent Ice/Snow zone having > 97% of its area on Status 1 and 2 lands. When vegetation zones are ranked by Conservation Priority Index (CPI), the four zones with highest priority based on low protection status, high conversion, and importance in terms of size, are three steppe zones (the Palouse, Big Sage/Fescue, and Wheatgrass/Fescue zones) and one westside zone (the Willamette Valley zone). Of the seven zones of moderately high CPI, four are steppe zones and three are the remaining Puget-Willamette Trough zones. Thus, seven of nine steppe zones and all four Puget-Willamette Trough zones have high or moderately high CPIs. Overall, 51% of the steppe zones has been converted to agriculture; 70-88% has been converted in the three steppe zones with the highest CPI. In the Puget-Willamette Trough zones (which encompass the major metropolitan areas of the state), 40-67% has been developed or converted to agriculture, and none of these zones have more than 15% of their area in conifer forest, the natural dominant cover.

Vertebrates

Distributions of terrestrial vertebrate species were modeled by intersecting range limits with suitable habitats (Fig. 1). We assigned codes to indicate habitat quality for each species based on ecoregion, vegetation zone, and land cover within the zone. Vegetation zones within an ecoregion were designated as "core" or "peripheral"; core zones were those in which the species was most common and peripheral zones were those in which the species occurred, but was rare or the zone was believed to be a population sink. Land cover was designated as "good," "adequate," or "contingently suitable" (i.e., suitable, contingent upon the availability of habitats below our minimum 100-ha mapping unit).

We assessed the protection status of vertebrates by: 1) calculating each species' total predicted distribution, the percentage of its distribution on Status 3 lands, and the percentage of its distribution of Status 1 or 2 lands; 2) mapping vertebrate species richness of various taxonomic groups and assemblages by overlaying predicted species' distributions; and 3) mapping areas of high vertebrate richness according to Conservation Status (Fig. 2). The effects of basing vertebrate richness analyses on presence/absence versus the most suitable habitats for each species were also explored. We found that presence/absence-based maps obscured the relative importance of low-elevation zones and habitats unaltered by human activity. All subsequent vertebrate analyses were based on the most suitable habitats for each species.

Amphibians: The number of native amphibian species is highest in mid- to late-seral conifer forests in low- to mid-elevation westside forest zones. Mid- to late- seral conifer forests in the Western Hemlock zone on the southern Olympic Peninsula and the southwestern Cascades have particularly high amphibian richness.

Reptiles: Native reptile richness is highest in the steppe zones and low-elevation eastside forest zones in steppe habitats, open forests, and forest openings.

Mammals: Habitats with high numbers of mammal species are riparian areas and forests in the Western Hemlock and Olympic Douglas-fir zones of the westside, and the Interior Western Hemlock, Interior Redcedar, and Grand Fir zones of the eastside, but the patterns of species richness vary greatly among mammalian subgroups.

Birds: Native bird richness is generally highest in low-elevation forests of the eastside and low-elevation wetlands throughout the State; however, the patterns of species richness varies considerably among avian subgroups.

We chose 10% representation on Status 1 or 2 lands to compare the relative protection status of taxonomic groups of vertebrates. For each group, the number of native species with less than 10% of their predicted distribution on Status 1 or 2 lands was:

Amphibians	14	of	24	(58%)
Reptiles	18	of	21	(86%)
Mammals	45	of	102	(44%)
Birds	138	of	230	(60%)

Other groups of interest included low-disturbance associates, state and federally listed species, and Columbia Basin-dependents. For these groups, the percentage of species with less than 10% of their predicted distributions on Status 1 or 2 lands varied between 38 and 100%.

For each species, we also calculated its total modeled distribution in Washington and the percentage of the modeled distribution on Status 1 or 2 lands. Though some caution must be used in comparing modeled areas between species at different trophic levels and in habitats of greatly differing productivity, our data do allow us to determine which species have a combination of low protection status and limited distribution, a warning sign of potential risk of extirpation.

Highest Conservation Priorities

Steppe zones and Columbia Basin-dependents: The most glaring gap in protection of biodiversity in Washington is in the steppe zones. The vegetation zones with the highest Conservation Priority Index (CPI) are steppe zones. Vertebrate species that rely on steppe usually have a correspondingly low percentage of their distribution on areas managed primarily for biodiversity.

Puget-Willamette Trough zones: These zones include the Puget Sound Douglas-fir, Woodland/Prairie Mosaic, Willamette Valley, and Cowlitz River zones. All have been heavily converted to both agriculture and development. The remaining forests are now a patchwork of hardwood, mixed, and early-seral conifer forest. There are only a few small areas of high richness of low-disturbance associates, as most of these species have been extirpated from these zones.

Ponderosa Pine and Oak Zones: These lowest elevation eastside forest zones have moderately high CPIs with less than 4% of their areas in Status 1 and 2 lands. They are zones of high reptile and avian diversity. Reduction in natural disturbance via fire suppression is a significant conservation problem in these zones.

Sitka Spruce and Western Hemlock Zones: These wet to mesic, westside forest zones have relatively little of their areas in development or agriculture, but logging has been extensive. They are zones of high amphibian and mammal (especially bat) richness, and their remaining mid- to late-seral forests support large numbers of amphibian, mammal, and bird species that adapt poorly to anthropogenic disturbance. Our data indicate that less than 8% of the Sitka Spruce zone and less than 10% of the Western Hemlock zone remain in late-seral forest; an additional 14% of the Sitka Spruce zone and 20% of the Western Hemlock zone were estimated to be in mid-seral forest.

Final Report Summary: Wyoming Gap Analysis Project

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The Wyoming Gap Analysis project (WY-GAP) recently completed its assessment of biological resources for the state. Our results show that less than 10% of the state of Wyoming is classified as Status 1 and 2 lands, and 90% of these lands occur in the Greater Yellowstone Area (GYA). Seven of the 41 land cover types occur at high elevations and are well (> 50%) protected in Wyoming because they occur in national parks and wilderness areas. Sixteen (44%) of 36 natural (nonanthropogenic) land cover types have < 1% or < 50,000 ha of the area they occupy in Status 1 and 2 lands. The highest priority for further protection is recommended for vegetated dunes, active dunes, forest-dominated riparian, shrub-dominated riparian, and grass-dominated wetlands because their current protection is low, and they are the most vulnerable to ongoing land management practices. However, wetland types are not satisfactorily mapped at our current MMU, and further efforts are needed to provide an adequate spatial description of their location before long-term planning for their conservation can be accomplished.

On average, a smaller percentage of the potential habitat of amphibians (8.8%) and reptiles (2.6%) occurs in Status 1 and 2 lands than either birds (14.4%) or mammals (14.5%). Species that have a high level of habitat protection (> 50%) were restricted to the GYA. Habitats of 6 (50 %) amphibians, 8 (31%) reptiles, 25 (22%) mammals, and 41 (14%) birds that are not considered peripheral in Wyoming merit increased management attention. The habitat of most of these species is primarily at low elevations in the eastern portion of the state or in the Green River area where Status 1 and 2 lands are uncommon. Management on multiple-use lands under the stewardship of the USFS in the Black Hills and the BLM in the Green River area, and cooperative efforts with private land owners in both the eastern portion of the state and in the Green River area, will be important to the long-term conservation of a large number of vertebrate gap species in Wyoming. However, we found that additional efforts to survey and map bat and rodent species will be necessary to reliably evaluate their current status.

For more information on the results of the Wyoming Gap Analysis, please obtain a digital copy of the report from the Wyoming Bioinformation Node web site at http://www.sdvc.uwyo.edu/wbn. If you have questions or would like a hard copy of the report, please contact Tom Kohley at (307) 766-2734 or kohley@uwyo.edu.

Final Report Summary: New York Aquatic GAP Pilot Project

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A pilot GAP project for aquatic systems began in 1995 to define a methodology and determine the feasibility of predicting biodiversity distribution. Similar to gap analysis in terrestrial environments, gap analysis for aquatic systems uses remotely sensed data for habitat mapping, infers aquatic biodiversity distribution from habitat data, and provides large-scale information for targeting conservation measures. Our pilot project has been a low-level effort (e.g., a one-person project) for two years. We established methods for database development and GIS analyses using one river basin in western New York.

The original purposes of gap analysis (Scott et al. 1993) remained unchanged when applied to aquatic environments. However, the connected nature of aquatic habitats and the mobility of aquatic species complicate traditional gap analysis methods. Emphasis was placed on streams and rivers, as these waterbodies harbor a large majority of the freshwater biodiversity in the United States and are the focus of water quality assessments by management agencies. Methods were developed to reflect habitat status over a network of streams because aquatic species respond to cumulative effects due to the flowing nature of water in streams. A key habitat attribute influencing aquatic species is water quality, thus we developed a nonpoint-source pollution model that relies on land cover. This component of our aquatic GAP model integrates the terrestrial and aquatic parts of gap analysis. Finally, aquatic biodiversity conservation will likely focus on land management, not land acquisitions, since aquatic biodiversity is generally highest in large streams and rivers, making land acquisitions impractical. Again, the linking of terrestrial and aquatic GAP coverages is essential to address conservation issues.

The basic aquatic GAP model predicts relative levels of fish and macroinvertebrate diversity and identifies stream reaches having high biodiversity that are without management or protection. This was accomplished by classifying stream segments into habitat types using five attributes: stream size, habitat quality, water quality, stream gradient, and riparian forest cover. Stream segments were classified into one of eighteen habitat types for fish diversity predictions and one of eight habitat types for macroinvertebrate diversity predictions. Fish species and macroinvertebrate taxa were linked to habitat types using life history data. Maps and information on management and conservation areas were included in the GIS to locate unprotected stream segments with high diversity. As in other GAP projects, these are the "gaps" or areas where future conservation efforts should be focused or management practices altered.

Our aquatic GAP pilot was developed for the Allegheny River watershed of western

New York. This region has a mix of forests (67%), agriculture (crop and dairy 28%), water and wetlands (3.4%), residential and urban areas (1.5%), and barren land (.1%). Aquatic habitats are largely comprised of small headwater streams (86% of the stream kilometers), with only 11% of the stream kilometers in large streams and small rivers and even fewer kilometers in large rivers. From our GAP model, 92% of the stream kilometers were modified or highly altered habitats leaving just 8% of the stream kilometers as potentially supporting the highest species diversity. The classification of stream segments into modified and highly altered categories was largely due to agricultural land use predominantly occurring in stream valleys and roads adjacent to streams. Approximately 79% of the stream kilometers were predicted to have good water quality using the nonpoint-source pollution model. When habitat status and water quality were combined, 913 (94%) of 980 stream segments were considered altered in some way. Thus, although degraded water quality was an important factor in the anticipated reduction of biodiversity for all sizes of streams, it was not nearly as dispersed or prevalent as physical habitat degradation. Overall high-quality stream segments were few and were well distributed across watersheds. These high-diversity habitats were the stream segments with high-quality water, intact channel habitat, a closed streamside canopy, and a high gradient. Good-quality streams were primarily headwaters (91%), with the remaining 9% comprised of large streams and small rivers.

The most diverse fish habitats were predicted to occur in large stream and small river segments with intact habitat quality and water quality suitable for life support. Only eight stream segments were identified in this class. Due to the large degree of human land use immediately adjacent to streams, there was an abundance of stream segments classified as modified and highly altered in habitat quality. This, in addition to the stream segments classified as degraded in water quality, greatly reduced the number of stream segments available for classification as highest in fish diversity. The most diverse macroinvertebrate habitats were predicted to occur in high-gradient, closed-canopy streams with good water quality (262 stream segments). Unlike in predictions for fish, the limiting factor in anticipated macroinvertebrate diversity was water quality degradation. Good water quality was predicted to be prevalent, especially in typically high-gradient headwaters, therefore there were many sites expected to have high macroinvertebrate diversity.

The goal of our pilot project was to demonstrate the feasibility and utility of the gap analysis methodology for predicting biodiversity distribution at the watershed scale. We illustrated this through the creation of a geographic information system model that classified stream habitats and related fish species and macroinvertebrate taxa to these habitats. The use of a land cover map in the prediction of water quality served as a link to gap analysis efforts in terrestrial systems. One major finding in this pilot project was the scarcity of stream segments with high predicted fish diversity, defined as large streams or small rivers with intact habitat and good water quality. GIS analyses showed that agricultural land use and roads were concentrated in midsized stream valleys where flat land along rivers was not vulnerable to destructive flooding.

Although intuitive as an afterthought, the GAP model identified this pattern and clearly explained why so few quality midsized streams remain in what is largely a forested setting.

Another finding suggests that the existing conservation status may not actually afford significant protection. Of all stream segments, about half (48%) were under some form of protection by state parks, wildlife management areas, state-regulated wetlands, and water quality management classes A or AA. Despite the large percentage of streams in protected areas, the conservationoriented management classes do not appear effective for aquatic biodiversity conservation. Many stream segments in protected areas were classified as poor quality. The protection of small land units did not substantially reduce the effects of runoff from agricultural lands or of alteration of streams along roads and farms. The utility of our GAP model in conservation planning was demonstrated by being practical to implement and capable of making predictions of the distribution of biodiversity and of management gaps.

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BIODIVERSITY PREDICTIONS: INTEGRATING URBAN GROWTH MODELS WITH LAND COVER DATA AND SPECIES HABITAT INFORMATION

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Introduction

Habitat loss and subsequent fragmentation due to urban development are part of a larger suite of anthropogenic impacts on biodiversity, but they now rank among the principal causes of species endangerment in the United States. Several types of urban growth simulation models have been developed which can supply useful information for biodiversity planning. In many cases, however, the data required for biodiversity planning may not be compatible with the urban models, leading to analytical inaccuracies and misleading conclusions. Here, I briefly introduce a case study for biodiversity analysis and examine several lines of logic likely to be employed in such assessments. I conclude with a discussion of assumptions built into the data and their influence on model outcome.

Techniques for Model Integration

Habitat quality and quantity aspects of biodiversity were examined using three principal inputs: urbanization scenarios, wildlife habitat maps, and species habitat models. Output from the analyses was reported as loss of habitat area or, in some cases, in terms of impact to the vertebrate species associated with degraded habitats.

A flow chart of the models and analyses provides an overview of the biodiversity sensitivity analysis (Figure 1).

Three different models for predicting patterns of urban expansion were tested. These included the 500-meter

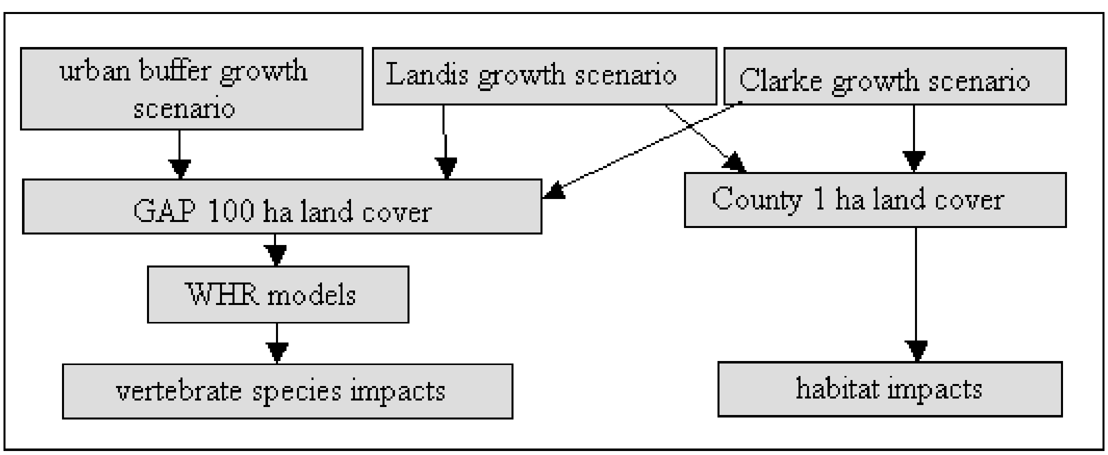

Figure 1. Flow chart for biodiversity sensitivity analysis. Three urban growth scenarios and two land cover models combine to evaluate vertebrate and habitat impacts in Santa Cruz County, California.

"urban buffer," "Landis" (Landis and Zhang 1998) , and "Clarke" (Clarke and Gaydos 1998) scenarios. Outputs from the different growth models were then used in conjunction with coarse-grain (100 ha minimum mapping unit) land cover maps from the California Gap Analysis Project (GAP, Davis et al. 1998) .

The Landis and Clarke models were also used with a finer-grain (1 ha) land cover data set. This map layer was commissioned by the Association of Monterey Bay Area Governments (AMBAG) based on 30-meter Landsat Thematic Mapper (TM) imagery. Spatial distributions of individual vertebrate species predicted to occur in the study area were made possible by applying wildlife habitat relationship (WHR) models (Airola 1988) to the coarser-grained GAP land cover data. Potential impacts of urban growth to these species were explored by intersecting scenarios of future urban growth from each of the three models with the WHR-based predicted distributions of the species (e.g., Figure 2).

Discussion

The species habitat analysis outlined here is a close examination of one major factor in the assessment of biodiversity. Other biodiversity elements such as ecoregional analysis, restoration potential, special features,

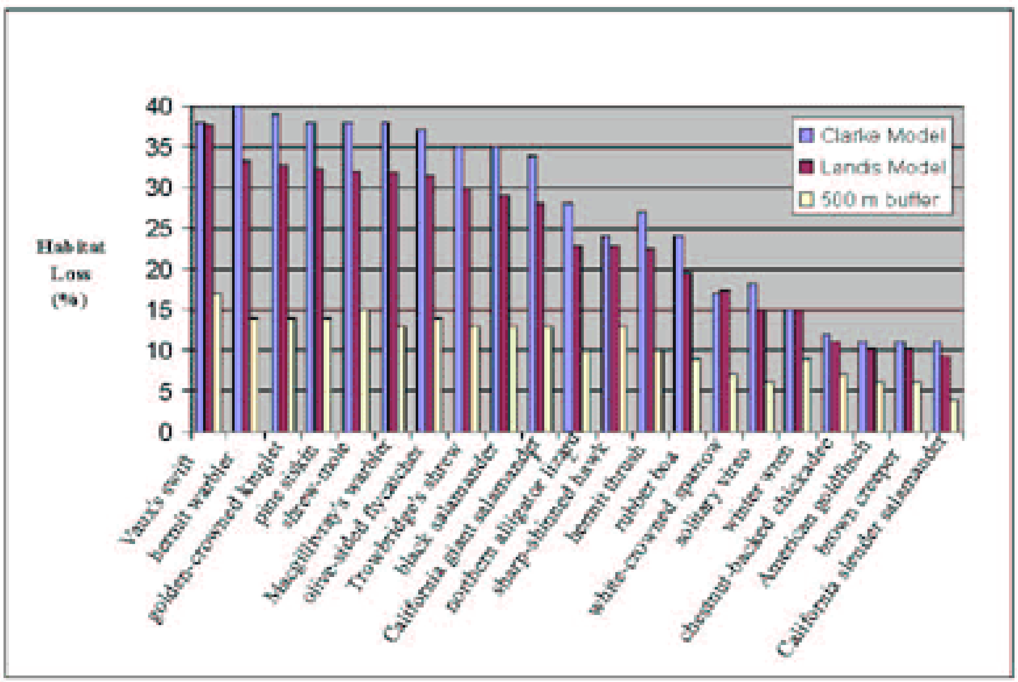

Figure 2. Comparison of predicted habitat loss under three growth scenarios in Santa Cruz County, California: 500-meter urban buffer, Landis growth model, and Clarke growth model. Species and habitat data are from the California Gap Analysis Project (GAP). Habitat classes are rank-ordered based on the results from the Landis model.

and habitat shape are also important (Cogan 2002) , though these were not specifically addressed in this study. The combination of urban growth models and land cover maps (Figure 1) was used to compare measures of habitat and vertebrate impacts. Here, habitat impacts were considered to be actual habitat areas converted to urban land use.

For example, if a 1,000 ha forest is reduced to 900 ha after urbanization, the habitat loss is 10%. If the same forest is reassessed in terms of native vertebrate habitat, it may be more important to consider buffer distances from impacts, non-linear predation effects, and other complex landscape metrics. These more specific approaches can be valuable in some instances; however, when applied to a regional study with many species, the results can be misleading. Stated differently, it is challenging to model disturbance effects as realistically as possible while working with a group of dissimilar species over a broad area.

The approach to vertebrate habitat assessment presented here assumed that if a highly intrusive land use such as urbanization entered a habitat patch, then the entire patch was likely to be compromised in terms of habitat quality for vertebrate species. In some instances, this assumption may have overemphasized the impact of urbanization. On the other hand, it was also likely that urbanization effects were underemphasized in cases where urban expansion approached (but not actually entered) a habitat area. An alternate model could employ spatial buffers to model the neighborhood effects of urbanization; however, this approach would introduce additional complexities, such as splitting map polygons, and imposes the need for species-specific analysis. Both the habitat and species types of impacts are important; however, it is necessary to clarify the conceptual differences between habitat and vertebrate impacts when evaluating or discussing urban growth impacts. The methods used in this analysis were based upon an underlying logical sequence most simply presented as a flow chart (Figure 3).

A central assumption here was that different urban growth patterns should have measurably different biodiversity impacts. As with any metamodel, it was also important to ensure that the data and various component models were compatible for integrated analysis. It is often illuminating to investigate where the logic of a scientific investigation might become unsound, as well as where it is strong.

The logical flowchart outlines key junctions where this type of biodiversity assessment might face impediments and offers explanations and recommendations for each situation.

Given perfectly accurate biodiversity and urban growth models, lack of biodiversity response will still occur if the two models are not spatially or thematically compatible. An indicator of this type of incompatibility can be seen in the comparison of vertebrate habitat losses following different

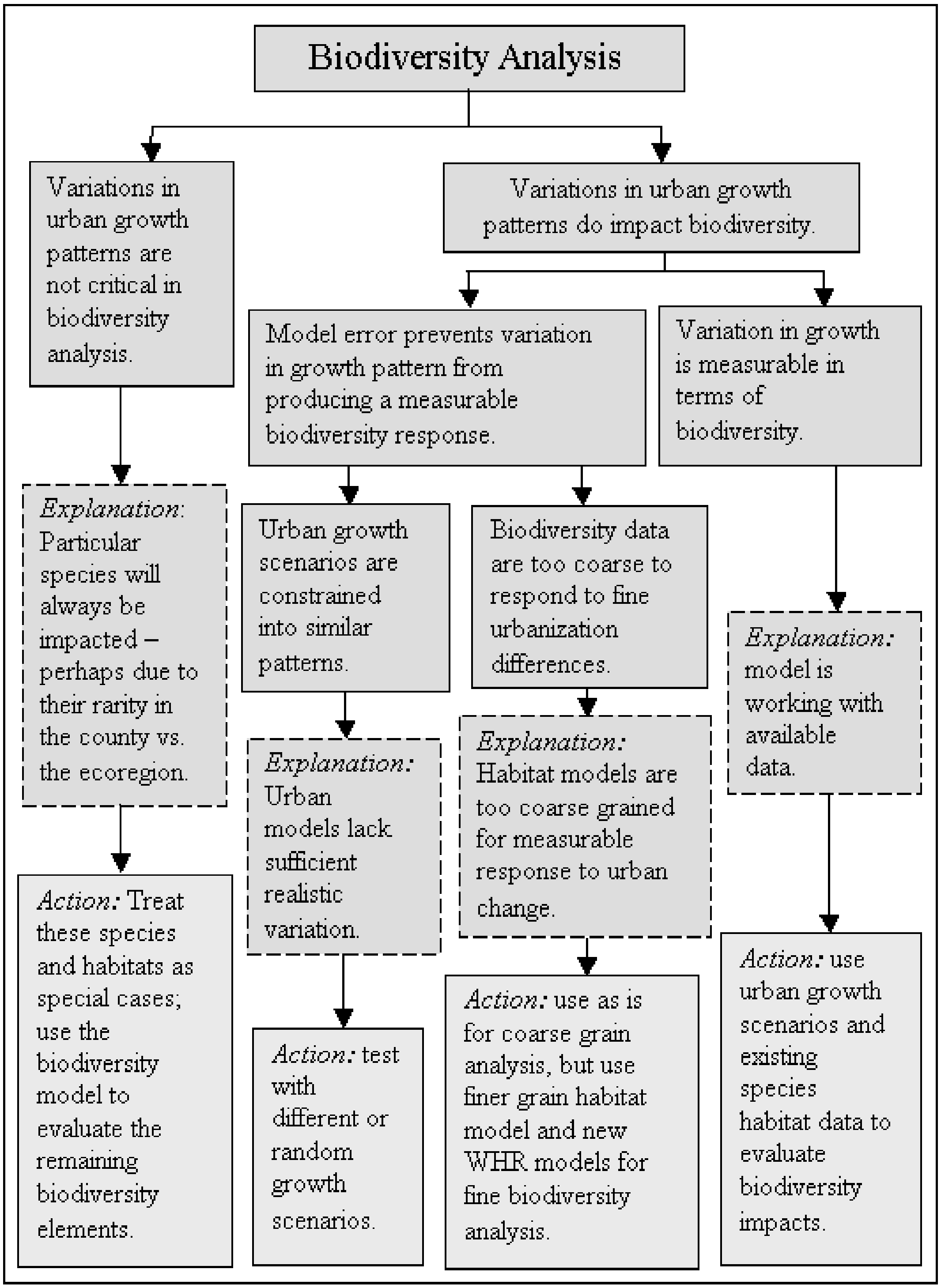

Figure 3. Logical flow chart for biodiversity analysis with urban growth models.

urbanization scenarios (Figure 2). One interpretation of this result suggests that vertebrate impacts are much the same following either the Clarke or the Landis models. Indeed, it seems remarkable that the *rank order* of species and even habitat impacts is so similar under two independent and seemingly different growth models. It would seem to require a radically different growth model like the simplistic 500-meter buffer to produce a significantly different outcome. Another, perhaps more likely, interpretation is also possible. If the GAP data on wildlife habitat relationships are spatially coarser that the growth models, our ability to differentiate between the Landis and Clarke models will be diminished. In support of this hypothesis, the appearance of the map products and (most importantly) the habitat impacts, indicated substantial differences between each of the three urban models.

The balance of spatial grain and thematic detail is an important consideration when producing and using maps of land cover for use in biodiversity analysis. Using the AMBAG 30-meter MMU land cover map, the fine map grain results in relatively large areas (up to 49,000 ha) to be mapped as contiguous albeit marginally connected patches. At slightly coarser map grains, many of the corridors of connecting habitat would merge into other classes, resulting in a very different data set for the habitat modeler. This example illustrates how fine-grain maps with coarse thematic detail can overemphasize habitat connectivity. In this case, the assumption that urban disturbance on the edge of a habitat patch impacts the entire patch becomes tenuous when using data with fine spatial grain but coarse thematic grain such as the AMBAG 30-meter land cover map. As 100-meter or finer-grain urban growth models gain acceptance as a reasonable spatial scale to model the biodiversity land use complex, more research is needed to ascertain the appropriate levels of thematic resolution in land use and land cover mapping.

There are several difficulties associated with measuring regional urban impacts on vertebrate species. The model presented here used polygons of habitat to represent potential distributions of vertebrate species and assumed that analysis of divided polygons was not a valid application of the data. Detailed studies of specific divided habitat polygons are possible, given appropriate species-specific data. However, this local approach will not be effective regionally. Urban development is sometimes seen as a continuous creeping of small steps, whereby each development project in isolation is difficult to assess for regional biodiversity impact. The species assessment method presented here used habitat polygons to model impacts, effectively dealing with the "urban creep" issue while maintaining biologically meaningful area units. The complementary combination of a discrete species metric (e.g., polygon-based) along with a continuous habitat model is a powerful and much needed approach.

As biodiversity models such as those discussed here evolve and build in complexity, our land cover maps and wildlife habitat relationship models will be pressed to deliver more information with higher quality standards. Some of our data sources have already evolved from simple maps of predicted species location to become temporally dynamic models of predicted species connectivity and spatial pattern. Unfortunately, most of our current maps are not up to this advanced standard. Like most modelers, cartographers have long known that the design constraints of producing the best habitat maps will depend on the specific questions being asked of the data. This fundamental principle is sometimes obscured or overlooked when we allow technological capabilities such as satellite sensor resolution and radiometric spectral response to overly influence our understanding of habitat classification and vertebrate distribution.

These findings were presented to facilitate an improved understanding of habitat and species impact models and to provide direction for future land use and land cover mapping. The specific models discussed here are important elements of more generalized biodiversity assessments, which are continually improving our understanding of biodiversity and promise to provide additional guidance to minimize the disruptive impacts of urbanization and development.

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A METHOD TO ASSESS RISK OF HABITAT LOSS TO DEVELOPMENT: A COLORADO CASE STUDY

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Introduction

Land use planning for private land is fundamentally important for conserving biodiversity nationwide (Dale et al. 2000). A major opportunity to refine the Gap Analysis methodology is to integrate socioeconomic factors to better assess both levels of protection and risk, particularly on private lands (McKendry and Machlis 1993). Incorporating information about private lands into the GAP methodology is important because private lands contain disproportionately high levels of biodiversity and habitat for rare species (Bean and Wilcove 1997); many of the important causes of habitat loss and habitat fragmentation stem from changes of land use on private lands; and they vary greatly in the degree of human-induced impacts on habitat.

GAP methodology identifies land cover types and species distributions that may be particularly vulnerable given their status in the current array of land ownership and management. However, a main drawback is that the coarse categories (4) of biodiversity management status, based on potential land use activities, may be weakly associated with actual species vulnerability (Stoms 2000). Some types of human activities cover broad expanses of the landscape and result in substantial land cover conversion, such as mono-crop agriculture and urban uses, and these activities typically are well-represented on land cover maps. However, land cover maps miss vast areas under the influence of either broad-extent, low-intensity land uses (e.g., low-density rural residential development) or small-extent, high-intensity activities such as oil and gas wells. Compiling data that more directly relate impacts on biodiversity associated with land uses is challenging (Stoms 2000), but offers a straightforward and reasonable means to identify threats to biodiversity, although actually demonstrating species responses to land use activities is quite challenging in practice (Theobald et al. 1997).

Another opportunity to refine status categories is to move beyond vulnerability and differentiate areas on the landscape (and species habitat) that are currently threatened or likely to be threatened in the future by land use activities associated with human development (e.g., urbanization, intensive agricultural practices, logging, etc.). Without considering these threats to species and habitat, conservation resources overall may not be properly prioritized (Cassidy et al. 2001) to achieve the greatest benefit for the most species (Scott et al. 1993). McKendry and Machlis (1993) described a general framework to extend biodiversity gap analysis by including socioeconomic indicators such as population change, economic trends, government policies, and land use conversion. Although current GAP methodology recognizes this limitation-for example, "We emphasize, however, that GAP only identifies private land as a single homogeneous category and does not differentiate individual private land units or owners…" (Csuti and Crist 2000)-few methods to address these limitations exist.

Recently, Stoms (2000) compared three indicators of development-permitted land use, "roadedness," and human population growth-to stewardship status for two pilot areas in California and found large differences between the more direct indicators and the general proxy of status or protection level. Theobald et al. (1998) developed a preliminary assessment methodology to examine the impacts of private land development on habitat using GAP land cover data, but did not quantify differences between management protection level and other indicators of land use.

Here we present an approach to refine the identification of vulnerable areas to consider what lands are threatened by various human land uses, especially those that have significant impacts and are increasing rapidly, such as urbanization and rural residential development. We utilized data readily available nationwide to develop a methodology to incorporate information about land use on private lands when assessing protection levels on private (and adjacent public) lands, and to forecast future levels of development to identify areas that are most at risk from potential private land development. We illustrate this approach using a case study from Colorado.

Colorado, often referred to as the "bellwether" of the Rocky Mountain West, has seen significant threat to habitat due to development pressures. Indeed, not only is the West's population growing three times as fast as the rest of the US (US Census Bureau 2001; Baron et al. 2000), but demographic and economic trends are changing the pattern and location of development (Riebsame et al. 1997). As a result, more than 60% of the West's counties are experiencing "rural sprawl," where rural areas (outside of city and town limits) are growing at a faster rate than urban areas (US Census Bureau 2001). In Colorado, population growth rates in nearly one-fifth of the counties exceeded 5% from 1990 to 1997, and this growth has caused large expanses of low-density development (Theobald 2000).

Methods

We developed two easily mapped measures of development and then used these indicators to assess which land cover types were particularly at risk and to identify where habitat is threatened by development. Our case-study assessment utilized both the land stewardship map and the species distribution maps produced by the Colorado Gap Analysis Project (Schrupp et al. 2001).

We selected two socioeconomic indicators to develop maps for and to test in relation to biodiversity: roads and housing density. The effects of roads on biodiversity and ecological integrity has been well documented (Forman and Alexander 1998). Road and housing density are often thought to be highly correlated, but because mixed results were obtained for a preliminary analysis (Theobald 1997), we chose to model both indicators to further test whether these were highly correlated for statewide areas. Although population density is often used to map human activity patterns, population data is tied to the primary place of residence and so underestimates potential effects on habitat in areas with a high percentage of second and vacation homes (Theobald 2000; Theobald *in press*).

Moreover, potential impacts to habitat such as removal of native vegetation, alteration of vegetation structure for defensible space for wildfire protection, and introduction of exotic species are more closely related to housing density.

Although road density is typically used as a measure of road effects on biodiversity, we created a "roadedness" map (Figure 1) following the methodology developed in California (Davis et al. 1996; Stoms 2000). Roadedness does not suffer from bias introduced when calculating road density in areas where many roads close together result in very high road densities and better accounts for spatial pattern.

Moreover, an important assumption in creating a map that depicts effects of roads on biodiversity is that larger roads (e.g., highways) typically affect species further from the road than smaller (e.g., local) roads, because larger roads are typically wider and carry more traffic. Therefore, the "roadedness" index estimates the proportion of an area (e.g., watershed, county, status category) that is affected by roads. Roads from US Census Bureau TIGER files were converted to 30 m GRIDs and then were assigned a buffer width according to the schedule in Table 1.

To map historical and current housing density, we used 1990 US Census Bureau block-groups and blocks, which are subdivisions of the familiar census tract. To account for underestimation of units in previous decades, decennial estimates for 1940-1980 were corrected using a correction factor

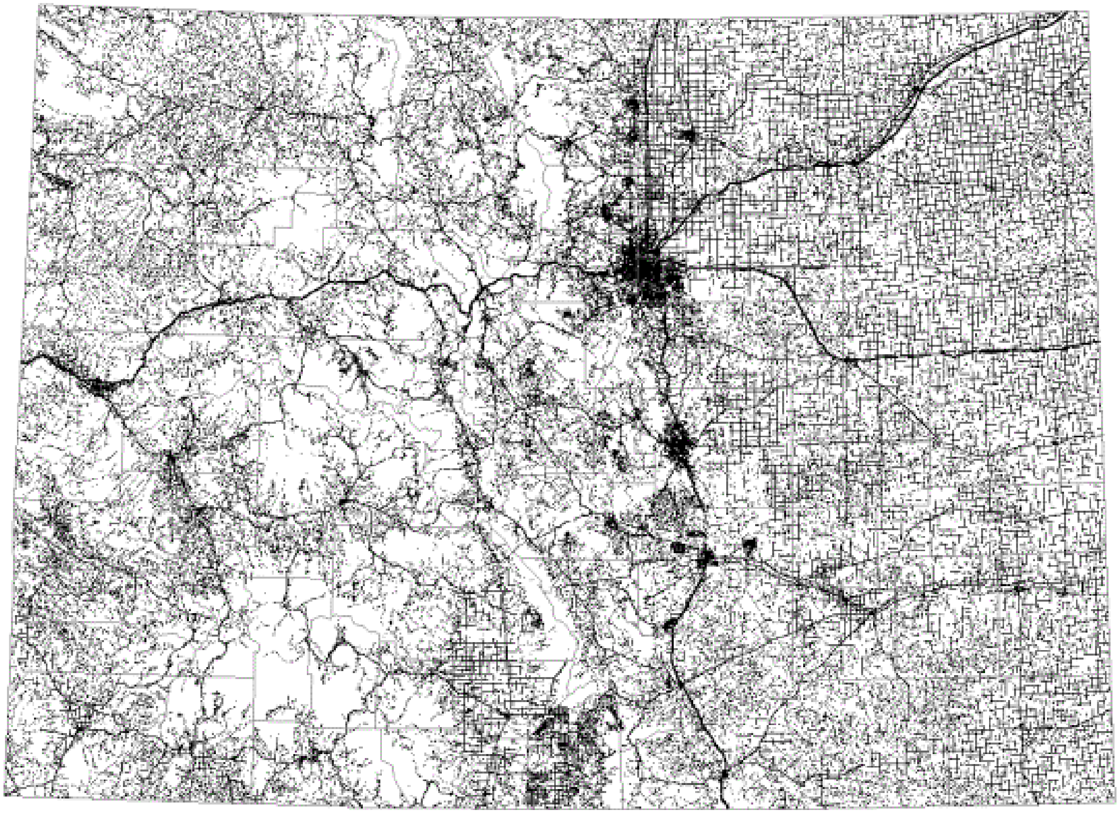


Figure 1. Roaded areas in Colorado.

Table 1. Roadedness index buffer widths. Total width of affected roaded portion is twice buffer width. After Davis et al. (1996) and Stoms (2000).

Census Feature Class Code	***Description***	***Road class***	***Buffer width(m)***	***Total width (actual)***	***Expand cells (30 m cell size)***
A10-A18	Primary (limited access or interstate highway)	1	500	1000 (990)	16
A20-A28	Primary (other US or State highway)	2	250	500 (510)	8
A30-A38	Secondary (state and county)	3	100	200 (210)	3
A40-A48	Local	4	100	200 (210)	3
A50-A58	Vehicular (4WD)	5	25	30	0
A70-A73	Other (hiking)	9	0	0	0

computed as the ratio of number of units in a county from historical census divided by total housing units summed from current estimates (Theobald 2001b). To map likely future housing density, we developed a model that recognizes and represents land use changes beyond the urban fringe (Figure 2). Although a number of approaches have been developed to forecast future growth patterns, most efforts have focused on *urban* growth and changes to urban or built-up cover types and are based on land cover types classified from satellite imagery and occasionally from high-altitude aerial photography (e.g., Brown et al. 2000). Recently, Clarke and Gaydos (1998) developed a California-based model to predict urban growth in San Francisco and Baltimore. Stoms (2000) distributed population growth using a rule-based approach that arbitrarily limited growth to 8 km expansion from urban cores.

Rather than rely on urban-centric models of housing growth, we used county-based population projections to derive the number of housing units needed in 2025 and 2050 (Theobald 2001a). We then spread these units throughout the block-groups by assuming that a block-group's density could not exceed the average housing density of its neighbors, for each decadal time step (Theobald et al. 2001).

We then analyzed the threats to habitat by overlaying the roadedness and housing density layers with land cover data.

Results

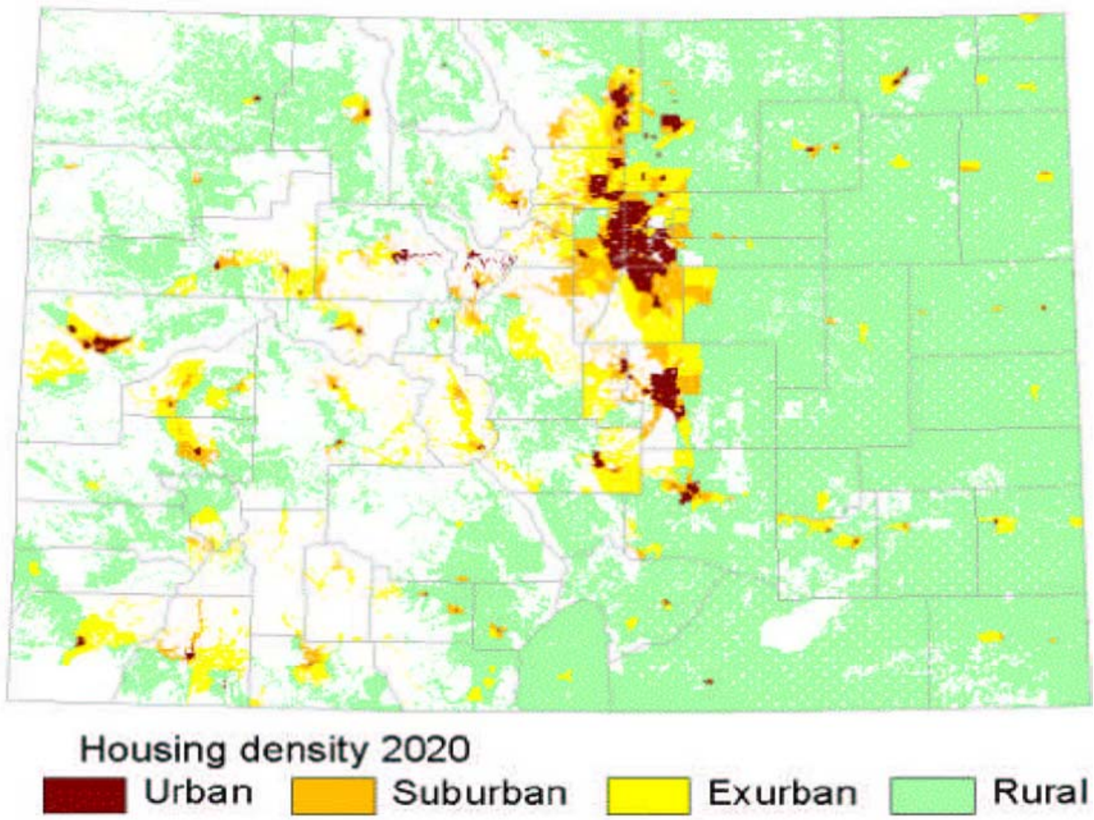
Over 269,000 kilometers (~167,000 miles) of roads were mapped in Colorado, resulting in 21.7% of Colorado being "roaded." Roaded proportion varies widely by watershed, from a low of 6.1% to a high of 40.9% (mean of 20.7%) (see Figure 3).

Contrary to common belief, there was a poor relationship ($R^2 = 0.21$) between percent roaded and the proportion of public land in each county. Although 10% of Colorado was "protected" (Status 1 and 2), about 13.5% of these protected areas were roaded. Conversely, the majority of Colorado was "unprotected" (Status 4), yet only about one-quarter of this area was roaded. About 5.1% of Colorado was developed in 1990 at densities higher than rural (i.e. urban, suburban, and exurban areas), and an additional 5% of Colorado will be "at risk" from new development forecasted for 2020, located mostly along the foothills of the Front Range and mountain valleys.

In Colorado, 24 of 43 natural land cover types were found to be *vulnerable*, which we define here as less than 10% protected in Status 1 and 2 (see Table 2). We designated a land cover class as *threatened* if 20% or more was roaded, or if 15% or more coincided with exurban or greater density development in 1990, was within 2 km of exurban or greater development in 1990, or coincided with areas at risk of development by 2020. Most vulnerable land cover types were also threatened by roads, although ponderosa pine, bristlecone pine, shrub-dominated wetland, and prostrate shrub/tundra were identified as threatened but were not identified as vulnerable. Tallgrass prairie, foothills/mountain grasslands, and bristlecone pine were identified as threatened by future development in 2020. Moreover, a number of land cover types proximal to development were found to be threatened, but were not identified as vulnerable, most notably water, spruce/fir, Douglas fir, ponderosa pine, bristlecone pine, forest-dominated wetland, and most tundra cover types.

Conclusion

Incorporating socioeconomic factors, such as road and housing density, provides an important opportunity to extend the methodology of gap analysis. We found that both road and housing density were useful indicators of potential impacts from activities associated with human land use and could be used to refine analyses of vulnerability to include level of threat (Figure 4). The data to produce these layers were readily available, and



Fugure 2. Housing density in 1990 and 2020.

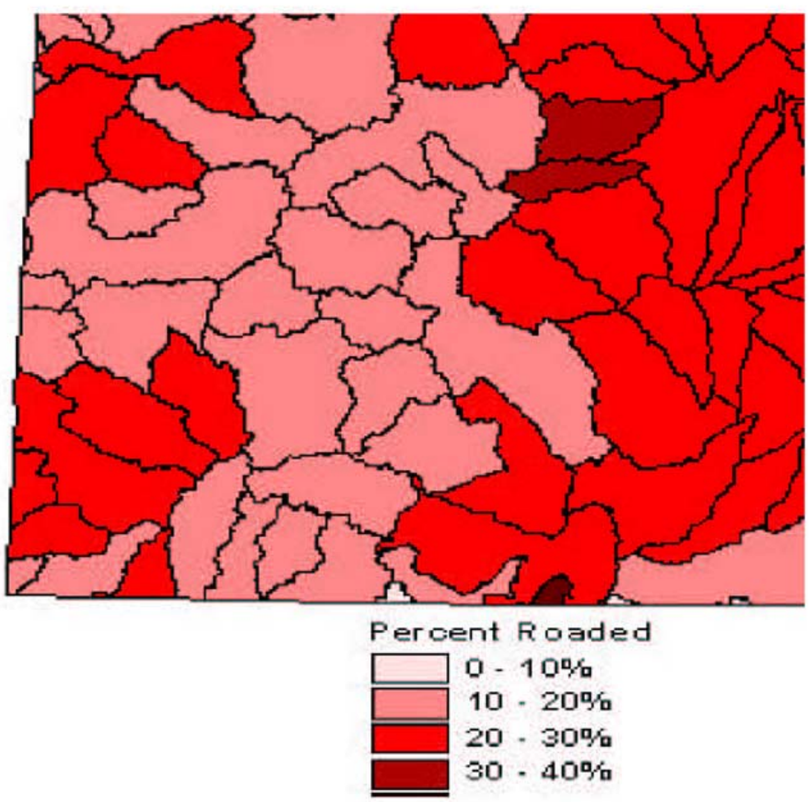


Figure 3. Percent roaded by watershed.

methods to convert them into reasonable indicators were straightforward. (Note: The derived maps of housing density are available at http://www.ndis.nrel.colostate.edu/davet/dev_patterns.htm).

In addition to roads and residential land use, there are a number of additional land uses associated with humans that would be useful but are more challenging to incorporate.

For example, additional data and methodologies are needed to better incorporate knowledge about the possible effects of grazing, logging, oil and gas wells, and fire suppression in spatially-explicit models of effects.

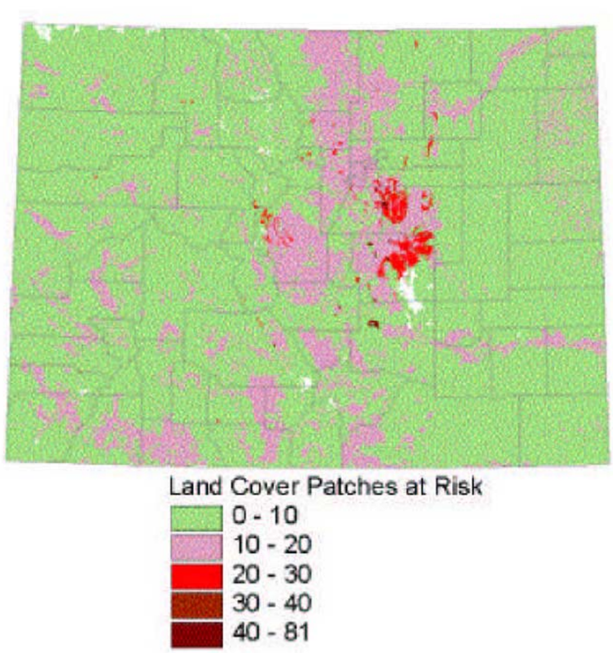


Figure 4. Patches of land cover ranked by percent "at risk" from development to 2020.

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Table 2. Statistics for proportion of protected, roaded, and developed for each land cover type in Colorado. Grey areas denote native land cover types that are 10% protected (Status 1 and 2), threatened by roads (>20%), or threatened by development (>15%).

*Land Cover (*human-made)*	*Class*	*Hectares*	*% of State*	*% Protected*	*% Roaded*	*% Developed in 1990*	*% w/in 1 km of developed*	*% w/in 2 km developed*	*% at risk of dev. in 2020*
Urban or built-up lands*	11001	217,270	0.81	0.19	84.44	88.4	95.3	97.2	13.4
Dryland crops*	21001	3,688,283	13.70	0.07	23.71	2.7	5.0	7.7	2.7
Irrigated crops*	21002	1,900,710	7.06	0.01	37.32	18.8	27.5	34.7	11.9
Orchards*	21003	222	0.00	0.00	29.73	98.7	100.0	100.0	80.6
Confined livestock feeding*	21004	458	0.00	0.00	45.41	48.7	48.7	48.7	-
Tallgrass prairie	31010	202,424	0.75	0.04	25.28	12.9	17.5	20.6	22.0
Sand dune grassland	31013	53,769	0.20	0.00	14.70	0.0	0.0	0.0	-
Midgrass prairie	31020	494,915	1.84	0.31	24.36	9.1	14.5	20.3	10.2
Shortgrass prairie	31030	4,029,190	14.96	0.19	23.14	1.1	2.7	4.4	1.2
Foothills/mountain grassland	31040	670,771	2.49	2.30	29.24	8.3	13.5	17.4	16.2
Mesic upland shrub	32001	116,051	0.43	3.26	22.86	11.8	21.3	27.0	11.1
Xeric upland shrub	32002	58,418	0.22	4.61	29.97	28.1	41.4	47.9	19.2
Gambel oak	32003	849,092	3.15	4.85	19.58	3.7	7.7	10.9	8.7
Bitterbrush shrub	32005	74,020	0.27	1.67	26.97	0.0	0.0	0.0	0.1
Mountain big sagebrush	32006	94,409	0.35	19.05	15.65	0.4	3.2	6.3	0.2
Wyoming big sagebrush	32007	44,364	0.16	0.00	24.03	0.0	0.0	0.1	-
Big sagebrush	32009	1,679,838	6.24	3.49	26.6	2.2	5.0	7.5	4.3
Desert shrub	32010	432,350	1.61	1.48	27.87	1.5	3.9	7.7	3.7
Saltbush shrub	32011	484,020	1.80	2.01	19.68	2.5	6.5	10.1	3.5
Greasewood fans and flats	32012	219,860	0.82	4.83	23.25	2.2	3.5	5.0	0.1
Sand dune shrub	32013	1,080,718	4.01	0.45	23.21	0.4	1.4	2.8	0.8
Disturbed shrub	32030	1,174	0.00	0.00	47.79	-	0.0	0.0	-

Spruce/fir	42001	1,871,967	6.95	46.53	9.14	1.5	9.5	16.8	1.6
Spruce/fir clearcut*	42002	9,200	0.03	8.38	29.68	0.0	0.0	0.0	-
Douglas fir	42003	432,356	1.61	14.13	14.69	7.1	24.1	34.3	7.0
Lodgepole pine	42004	872,309	3.24	34.44	15.31	6.6	16.0	20.9	4.1
Lodgepole pine clearcut*	42007	16,245	0.06	5.74	26.51	0.3	3.7	3.8	-
Limber pine	42009	1,227	0.00	0.08	18.34	0.0	0.0	0.4	-
Ponderosa pine	42010	1,388,349	5.16	12.68	20.96	13.7	28.2	34.8	10.7
Blue spruce	42011	2,940	0.01	46.53	2.79	0.0	0.0	0.0	-
White fir	42012	4,012	0.01	0.00	26.99	0.0	0.0	0.0	-
Juniper woodland	42015	466,417	1.73	12.16	15.34	0.3	1.2	2.7	1.4

Pinyon juniper	42016	2,503,871	9.30	7.24	17.93	1.9	6.4	9.9	4.2
Bristlecone pine	42017	22,813	0.08	10.31	28.85	14.8	30.4	38.0	26.5
Mixed conifer	42018	183,212	0.68	24.19	15.11	2.1	7.9	13.5	0.3
Mixed forest	43000	83,117	0.31	16.25	15.70	0.8	4.8	7.9	1.7
Open water	52001	90,794	0.34	13.47	16.69	6.4	28.1	37.0	3.9
Forest dominated wetland/riparian	61001	114,414	0.42	9.16	27.79	11.5	27.2	33.9	6.8
Shrub dominated wetland/riparian	62001	52,217	0.19	13.77	21.38	5.3	10.2	13.1	3.5
Graminoid and forb dominated wetlands	62002	45,468	0.17	6.70	27.87	2.9	7.6	10.5	6.3
Barren lands	70000	16,950	0.06	1.74	56.45	54.4	72.2	83.2	40.7

Unvegetated playa	71001	388	0.00	0.00	8.76	0.0	0.0	0.0	-
Sandy areas other than beaches	73000	18,054	0.07	0.00	13.98	0.6	1.4	2.8	-
Exposed rock*	74001	46,072	0.17	50.78	4.22	1.0	6.4	10.8	1.2
Mining operations*	75001	6,916	0.03	1.13	8.66	24.7	41.8	49.7	23.9
Prostrate shrub and tundra	81001	127,132	0.47	74.53	44.66	1.5	9.2	15.9	1.9
Meadow tundra	82001	183,496	0.68	62.92	2.64	1.8	16.6	27.9	1.0
Subalpine meadow	82002	204,731	0.76	28.28	4.50	4.8	14.1	21.3	3.8
Bare ground tundra	83000	200,106	0.74	81.59	18.33	2.1	13.1	21.3	2.0
Mixed tundra	85000	299,941	1.11	66.47	0.92	1.3	13.2	22.5	2.9

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