Section I. Wyoming Basin Rapid Ecoregional Assessment Overview and Synthesis

Chapter 2. Assessment Framework

By Natasha B. Carr, Kirk R. Sherrill, Annika W. Walters, Steven L. Garman, Jeffrey Wesner, and Jena R. Hickey

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Landscape Intactness

One of the primary goals of the Rapid Ecoregional Assessment (REA) is to identify areas that have high conservation potential, also referred to as "large intact areas." Intactness has been defined by the Bureau of Land Management (BLM) as "a quantifiable estimate of naturalness measured on a gradient of anthropogenic influence and based on available spatial data." At the ecoregion level, the ecological value of large intact areas is based on the assumption that because these areas have not been greatly altered by human activities (such as development), they are more likely to contain a variety of plant and animal communities and to be resilient and resistant to changes resulting from natural disturbances such as fire, insect outbreaks, and disease (Peterson and others, 1998; Gunderson, 2000). Large intact areas may be more likely to support viable populations of species and facilitate seasonal movements and dispersal of organisms. In addition, the potential for maintaining ecological processes across a variety of temporal and spatial scales is expected to be greater in larger intact areas. The concept of landscape intactness provides the foundation for the Wyoming Basin REA assessment framework, which quantifies the degree to which landscapes have been altered by development and other anthropogenic influences.

Rapid Ecoregional Assessment Components

Management Questions

Core and Integrated Management Questions form the basis for the REA assessment framework and were addressed for each species and community.

Core Management Questions

- Where is the Conservation Element, and what and where are its key ecological attributes?
- What and where are the Change Agents?
- How do the Change Agents affect the key ecological attributes?

Integrated Management Questions

- Where are the areas with high landscape-level ecological values (based on key ecological attributes)?
- Where are the areas with high landscape-level risks (based on Change Agents)?
- Where are the areas with high conservation potential (highest values, lowest risks)?
- Where are the potential areas for restoration (highest values, moderate-to-high risks)?
- Where are the potential areas for development (lowest values, highest risks)?

Conservation Elements

Conservation Elements represent the regionally significant species and ecological communities of management concern. The emphasis on ecological communities is based on the premise that intact and functioning ecological systems are more resistant to both natural and human stressors and more resilient to these agents of change (Noss 1987; Poiani and others, 2000; Parrish and others, 2003). Because it is not feasible to manage or monitor all species individually, protection of intact ecological communities may help to serve as a safety net for species not addressed specifically by the REA.

Priority species or species assemblages of management concern, which may not be adequately addressed at the community level, were specifically evaluated as Conservation Elements.

Ecological Communities

The seven ecological communities evaluated for the Wyoming Basin REA are streams and rivers, wetlands, riparian forests and shrublands, sagebrush steppe, desert shrublands, foothill shrublands and woodlands, and montane/subalpine forests and alpine zones. The initial selection process is described in the Wyoming Basin REA work plan (Carr and others, 2013). Ecological communities were chosen to represent the dominant species and life forms (also referred to as biomes) to ensure that the communities were not defined too narrowly. Fine-scale plant communities are difficult to map accurately at regional scales using national datasets, such as individual LANDFIRE Existing Vegetation Types. In addition, the use of the ecological communities is useful for addressing disturbance regimes and other broad-scale processes, including climate change.

Species and Assemblages

We evaluated a total of 14 species and species assemblages (aspen forests and woodlands, five-needle pine forests and woodlands, juniper woodlands, cutthroat trout, three-fish assemblage, northern leatherside chub, sauger, spadefoot assemblage, greater sage-grouse, golden eagle, ferruginous hawk, sagebrush-obligate birds, pygmy rabbit, and mule deer) as Conservation Elements (see 1–3 in Chapter 1—Introduction and Overview for a full list of species in assemblages). To be included in the REA, species or assemblages needed to meet criteria I—III and IV or V (criteria I−V listed below).

I. Regionally significant species or communities—Occurrence throughout the jurisdiction of at least three BLM Field Offices, with an emphasis on widely distributed species.

II. Species directly tied to management priorities and issues.

III. Species not represented adequately by ecological communities or other species.

IV. Species of conservation concern or assemblages as determined by BLM and other state and federal agencies.

V. Commodity species (game or furbearer species) (Knick and others, 2011).

Key Ecological Attributes

Key ecological attributes are fundamental characteristics of species and communities that contribute to their long-term persistence and resilience to Change Agents. We classified key ecological attributes into three categories: (1) biophysical characteristics and ecological processes that regulate the occurrence (distribution and abundance), (2) landscape structure (patch sizes and structural connectivity), and (3) landscape dynamics (natural disturbances or hydrological regimes) of species and ecological communities. Key ecological attributes and associated indicators were identified for each species and summarized in tables (see tables 2–1 and 2–2 for example formats of key ecological attribute tables provided in chapters of Sections III and IV). Elements of landscape structure include patch size and spatial distribution of patches, patch characteristics (such as core area), structural connectivity based on spatial distribution of patches, and the characteristics of the matrix between patches (Wiens, 2002; Wiens and others, 2002). We summarized the Change Agents that were evaluated and associated indicators (see tables 2–3 and 2–4 for examples of Change Agent tables provided in each chapter of Sections III and IV).

Table 2–1. Example of the used to summarize key ecological attributes[1] and associated indicators for terrestrial Conservation Elements for the Wyoming Basin Rapid Ecoregional Assessment.

Attributes	Variables	Indicators
Amount and distribution	Total area	Distribution based on LANDFIRE, species distribution models, or mapped occurrences.
Landscape Structure	Patch size	Patch-size frequency distribution.
	Structural connectivity	Interpatch distance that provides an index of structural connectivity for baseline distribution maps at local, landscape, and regional levels.
Landscape dynamics	Fire occurrence	Locations of fires and annual area burned since 1980.

[1] Key ecological attributes are evaluated for baseline conditions. Baseline conditions are used as a benchmark to evaluate changes in the total area and landscape structure of the distribution map for terrestrial species and communities due to Change Agents. Baseline conditions are defined as the potential current distribution without explicit inclusion of Change Agents.

Table 2–2. Example of the used to summarize the key ecological attributes and associated indicators of aquatic Conservation Elements for the Wyoming Basin Rapid Ecoregional Assessment.

Attributes	Variables	Indicators
Amount and distribution	Total length of streams and rivers by hydroperiod	Distribution based on U.S. Geological Survey National Hydrography Dataset
Landscape Structure	Stream length by watershed	Total length of streams by sixth-level watershed
Landscape dynamics	Recent fire occurrence	Proportion of sixth-level watershed burned since 1980
	Hydrological regime	Mean summer flow and timing of peak flows

[1] Key ecological attributes are evaluated for baseline conditions. Baseline conditions are used as a benchmark to evaluate changes in the amount and landscape structure of the distribution map for aquatic species and communities due to Change Agents. Baseline conditions are defined as the current distribution without explicit inclusion of Change Agents.

Table 2–3. Example of the used to summarize the Change Agents and associated indicators influencing terrestrial Conservation Elements for the Wyoming Basin Rapid Ecoregional Assessment.

Change Agents	Variables	Indicators
Development	Terrestrial Development Index	Percent of distribution in seven development classes
		Patch-size frequency distribution that is relatively undeveloped or has a low development score compared to baseline conditions
		Interpatch distances that provide an index of structural connectedness for relatively undeveloped areas at local, landscape, and regional levels
Climate change	Projected temperature and precipitation	Potential distribution of biomes based on projected bioclimatic envelope in 2030

Table 2–4. Example of the used to summarize anthropogenic Change Agents and associated indicators influencing aquatic Conservation Elements for the Wyoming Basin Rapid Ecoregional Assessment.

Change Agents	Variables	Indicators
Development	Aquatic Development Index	Percent of streams and rivers in each of seven development classes
		Length of streams and rivers that are relatively undeveloped or have low development compared to baseline conditions
	Barriers affecting patch size and structural connectivity	Number of dams and potential barriers (diversions and road crossings)
Invasive species	Presence and expansion risk of tamarisk and Russian olive	See Chapter 10—Riparian Forests and Shrublands
Climate change	Hydrologic regime change	Projected mean summer flow and timing of flow

Change Agents

We evaluated the four primary Change Agents required for the REA (development, fire, invasive species, and climate change). It is important to note that fire and climate (for example, drought) are inherent drivers of ecosystem dynamics in the Wyoming Basin, but fire and climatic regimes may be influenced by human activities (Rowland and Leu, 2011). In turn, human alteration of natural disturbance regimes can lead to habitat loss and other negative effects on species and species assemblages.

Ecological Conceptual Models

For each species, we also summarized the information provided in the key ecological attribute and Change Agent tables (tables 2–1 to 2–4) in the form of an ecological conceptual model or diagram (fig. 2–1). These diagrams are intended to provide visual representation of some of the primary potential interactions and feedback among the drivers and stressors (Change Agents) that were evaluated for each species and community. We used a standard format so that key ecological attributes and Change Agents that were not addressed (either due to limited data availability sufficient for regional-scale analyses or because the factor was not expected to be a major issue) are readily apparent. These diagrams are intended to highlight factors relevant to the REA and were not an exhaustive synthesis of all factors important to a species or community.

Assessment Framework

The assessment framework is comprised of a standard suite of approaches for evaluating landscape-level status based on Management Questions for each species and community (fig. 2–2). We used a similar approach to address landscape intactness for the Wyoming Basin ecoregion overall (Chapter 29—Landscape Intactness). The assessment framework was developed after reviewing other REAs and ecoregional assessments (including Quigley and Arbelbide, 1997; Hanser and Knick, 2011), and broad-scale assessment methods (including Parrish and others, 2003; Leu and others 2008; Annis and others, 2010; Theobald, 2010; Wade and others, 2011). We adapted methods that best addressed the overall objectives of the REAs and Management Questions, in addition to developing some new approaches to address landscape-level intactness.

Scale

Scale is an important consideration for conducting REAs. The resolution of the derived data, the appropriate scales used for summarizing or analyzing data, and the scale of reporting units can vary and have important effects on the results and conclusions (table 2–5). In some cases, one analysis scale may be sufficient to summarize information for a particular Management Question, but in other cases (such as for evaluating structural connectivity) multiple scales of analysis may be necessary to summarize information for patterns that vary dramatically across spatial scales (Noss, 1990). Because habitat patches may be hierarchically structured (for example, foraging patches may nested within breeding territories), decisions at a particular scale (such as where to forage) may be constrained by decisions at broader spatial scales (such as where to establish a territory) (Kotliar and Wiens, 1990). In addition, the effects of development can vary by temporal (fig. 2–3) and spatial scales (fig. 2–4; table 2–5).

We define three nested levels for evaluating terrestrial species and communities in the REA: local, landscape, and regional. The local level corresponds to patterns and processes that may occur over distances often less than a kilometer (km) or mile (mi). The local level also corresponds to the scale of field-level data, such as monitoring restoration success or quantifying tree density. Because of the broad-scale nature of regional datasets, detailed information at the local level generally is not available for the entire ecoregion. The landscape level corresponds to scales at which most analyses are conducted for this REA, generally a few kilometers or miles. Regional levels correspond to patterns that occur across large portions of the ecoregion, including the entire region. We used three nested watershed levels for evaluating aquatic species and communities: catchment (defined using GIS), sixth-level watershed, and fifth-level watershed (Seaber and others, 1987).

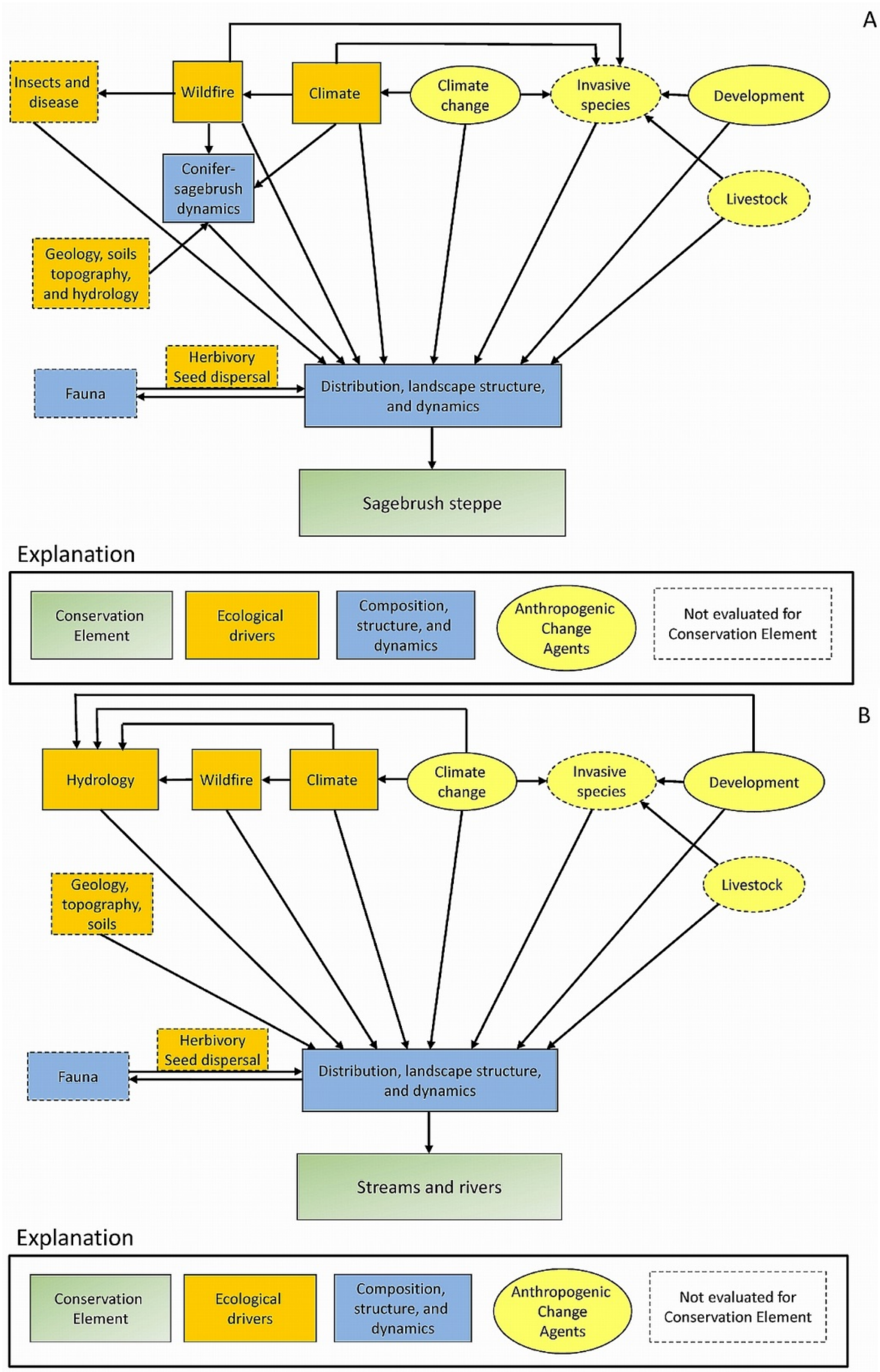

Figure 2–1. Examples of the generalized ecological conceptual model used for the Wyoming Basin Rapid Ecoregional Assessment (REA). (*A*) Conceptual model for sagebrush steppe, and (*B*) conceptual model for streams and rivers. Biophysical attributes and ecological processes regulating the structure and dynamics of ecological communities or species' habitat are shown in orange rectangles; additional ecological attributes are shown in blue rectangles; and key anthropogenic Change Agents that affect key ecological attributes are shown in yellow ovals. The dashed lines indicate components not addressed by the REA.

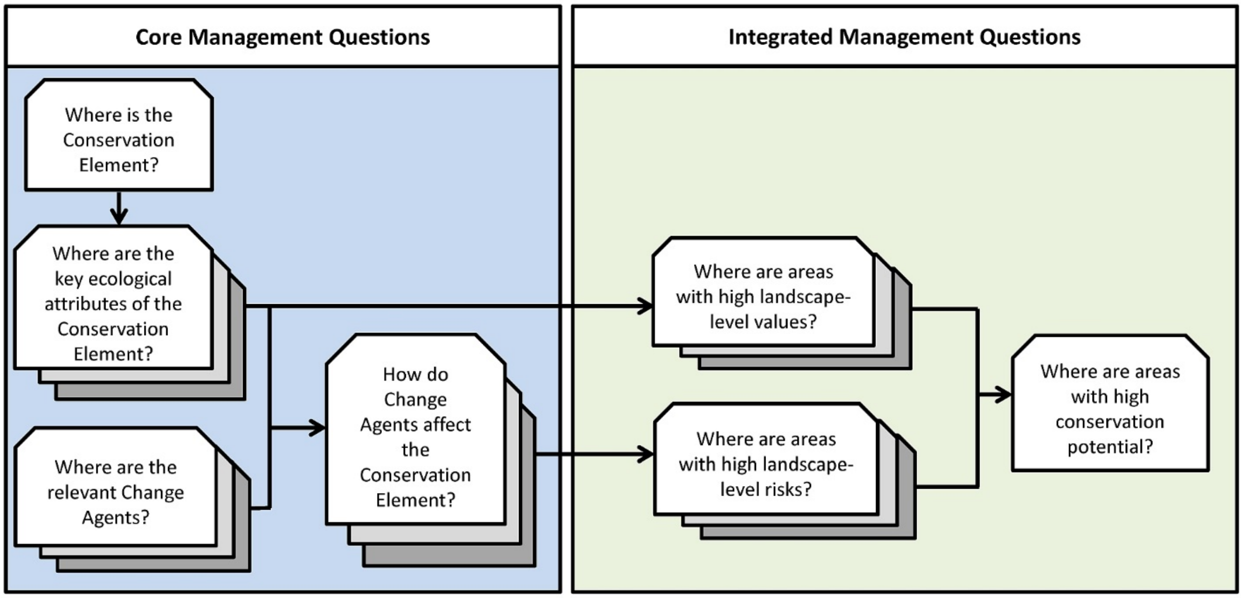

Figure 2–2. Generalized framework and approach for assessing the status of Conservation Elements and addressing Core and Integrated Management Questions.

Table 2–5. Relationships among levels or scales of ecological processes, administrative units relevant to agency decisionmaking, Rapid Ecoregional Assessment metrics and analysis units, and ancillary data sources.

	Level			
	Local	Landscape	Regional	West-wide
Bureau of Land Management administrative units	Field Office	Field Office, District Office	Multiple districts, State Office	Multiple State Offices, National Office
Project, permit, lease, plan, policy	Environmental Assessments, Categorical Exclusion, Range Allotment Evaluation	Landscape-level projects, Environmental Impact Statements, Environmental Assessment of cumulative effects, Resource Management Plans	Statewide Resource Management Plan Amendments, Environmental Impact Statements and Resource Management Plan cumulative effects	2005 Energy Policy Act, multistate projects
Ecological patterns and processes	Nest-site selection, foraging patch, home range, lek attendance, population density, community composition, patch size and connectivity	Home range, seasonal movement, dispersal and migration, population dynamics, habitat dynamics, disturbance regimes, patch complex sizes and connectivity	Seasonal movement, dispersal and migration, gene flow, range shifts, population dynamics, habitat dynamics, disturbance regimes, patch complex size and connectivity	Migration, range shifts, population dynamics, gene flow, disturbance regimes, connectivity
Rapid Ecoregional Assessment metrics	Relatively undeveloped patches, patch connectivity	Patch connectivity, landscape intactness, landscape-level ecological values and risks, conservation potential	Patch connectivity, landscape intactness, Rapid Ecoregional Assessment, landscape-level ecological values and risks, conservation potential	West-wide map of landscape intactness
Rapid Ecoregional Assessment analysis units	30-meter LANDFIRE, development footprint, catchments, local-level connectivity	2.25-kilometer moving window, township, sixth-level watershed, landscape-level connectivity	Township, fifth-level watershed, regional-level connectivity	Multiple states, multiple ecoregions, fourth-level watershed
Ancillary data for “step down’” assessments	Range inventory, permanent transects, Forest inventory, wildlife habitat surveys	Range inventory, permanent transects, forest inventory, wildlife habitat surveys	Forest Inventory Analysis, Western Governors’ Association Crucial Habitat Assessment Tools	Forest Inventory Analysis, Crucial Habitat Assessment Tools

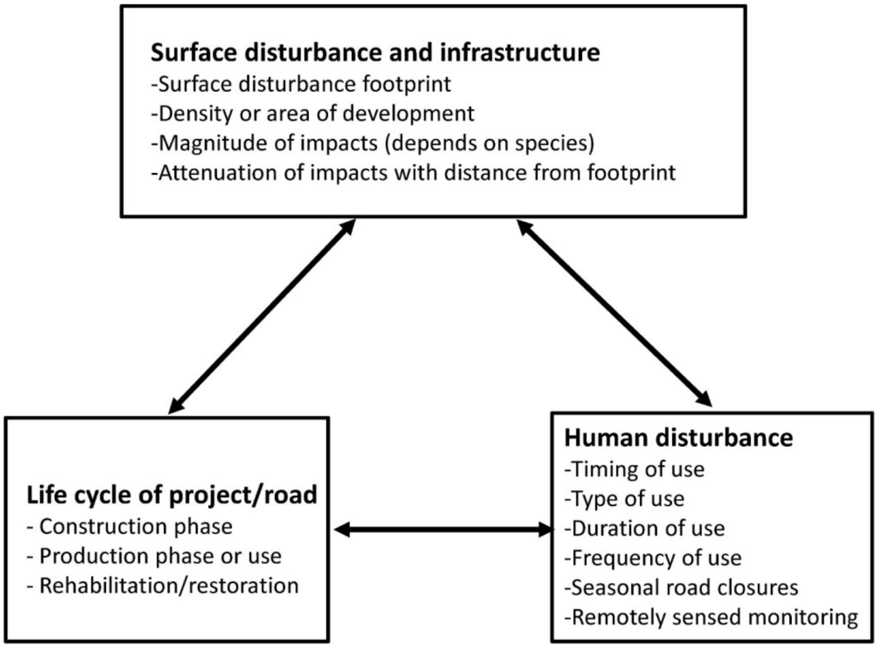

Figure 2–3. Temporal and spatial variation in the potential effects of development on species and communities. Management practices such as rehabilitation and restoration, seasonal restrictions, no surface occupancy, and use of remotely sensed monitoring of well pads to limit traffic can help to lessen the effects of development.

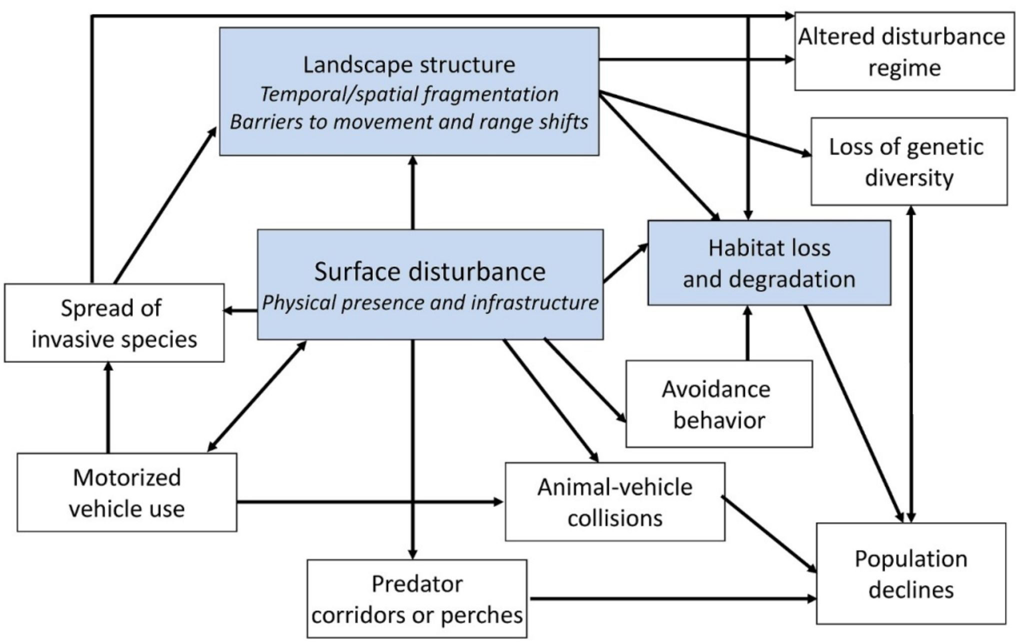

Figure 2–4. Potential cumulative effects of development on sensitive wildlife species. Blue boxes highlight the primary effects of development addressed by the Wyoming Basin Rapid Ecoregional Assessment, including surface disturbance, landscape structure, and potential for habitat loss and degradation. Other human activities associated with development and indirect effects of development could not be evaluated at the ecoregion level due to the lack of regional data but could be addressed at local levels.

Baseline Conditions

We defined baseline conditions as the potential current distribution of species and communities without explicit inclusion of Change Agents (the methods used for developing distribution maps vary and are described in the individual chapters; Sections III and IV). Thus, baseline conditions were used as a benchmark to evaluate the potential alteration of landscape-level features (including distribution, patch size, and structural connectivity) by Change Agents. For terrestrial systems, baseline conditions differ from reference conditions to an unknown degree as a consequence of previous vegetation conversion, which cannot be determined from regional or national data sources (such as LANDFIRE, 2010) used to develop baseline distribution maps. This limits our ability to quantify fully the direct effects of past vegetation conversion by development at a local scale (for example, the effects of roads on core area of aspen). However, evaluating development using relatively large window sizes (such as 16 square kilometers [km^2], see Terrestrial Development Index below) allows us to capture current and previous vegetation conversion due to development, including agriculture, roads, railroads, well pads, and mines, that could influence a particular cell. This approach, however, does not capture conversion from one type of nondeveloped vegetation community type to another, but we assume that this type of conversion is more likely in the vicinity of other anthropogenic modifications of the landscape.

Aquatic species and communities have similar issues relating to past vegetation conversion. Additionally, many of the fish species have greatly altered distributions compared to historic ranges. Habitat loss, altered flows, and dams or diversions leading to decreased connectivity of habitats has led, in many cases, to extirpation of some fish populations in portions of their historic ranges, such as cutthroat trout and sauger. Furthermore, some of the occurrence data we obtained from state agencies define populations based on the locations of barriers, such as dams and natural breaks, and consequently loss of connectivity due to development cannot be fully accounted by our approach. Some wetlands are created by agricultural activities, further compounding this problem. Despite these challenges, quantifying development at the broad scales evaluated here (see below) is a useful technique for assessing the current landscape-level status of the species and communities, with the caveat that all previous anthropogenic changes cannot be fully represented with available region-wide data.

Amount and Distribution of Conservation Element

Distribution maps for each species and community were based on LANDFIRE, mapped occurrences, available species distribution models (such as that of Hanser and Knick [2011]), or species distribution models developed as part of the REA. These were used to evaluate key ecological attributes (tables 2–1 and 2–2) and are briefly described in the relevant chapters in Section III—Communities and Section IV—Species (see Distribution Mapping of Communities and Species section in the Appendix for additional details).

Development

Because land uses can affect terrestrial and aquatic ecosystems differently, we evaluated development for terrestrial and aquatic systems separately.

Terrestrial Development Index

The Terrestrial Development Index (TDI) quantifies levels of development intensity, including agriculture, roads and railroads, energy and minerals, transmission lines, and urban development. The primary variables associated with terrestrial development (table 2–6) were compiled into the overall development index. To facilitate compilation of the development variables, we used a common metric, surface disturbance, to quantify each variable. The TDI is based on the percent of surface disturbance footprint for all terrestrial development variables in a 16-km^2 (6.18-square-mile [mi^2]) moving window. The moving window analysis uses the ArcGIS focal neighborhood statistics function (Environmental Systems Research Institute, 2011). The size of the moving window captures broad-scale cumulative effects of development that cannot be determined at much smaller scales. The TDI is based on Leu and others (2008) and Theobald (2010) but differs in many key respects (see Change Agents: current conditions in the Appendix for additional details on TDI methods).

The TDI scores range from 0 to 100 percent and were divided into seven classes for visualization purposes (figs. 2–5 and 2–6). The TDI scores ≤3 percent, which correspond to a frequently used management target for development levels, were separated into 1 percent increments, as requested by the REA Assessment Management Team. The TDI scores between 0–1 percent represent areas with few roads and (or) a very low density of oil and gas wells (fig. 2–5). The TDI scores between 1–3 percent often include low densities of oil and gas wells and roads (fig. 2–5B), whereas development index scores above 3 percent represent moderate-to-high levels of development, including relatively large oil and gas fields, surface mines, agricultural fields, centers of urban development, and highway/interstate corridors (fig. 2–5A, C).

Because TDI scores are continuous, alternative classes can be used to display the data to address a particular management question. The breakpoints used for the REA were derived from integer values corresponding to 10 equal subsets of the data (quantiles). For the TDI classes <5 percent, the classes compiled quantile break points with <1 percent increments. For the TDI classes between 5 and 20 percent, 2 quantiles were compiled (see Selection of Terrestrial Development Index breakpoints in Appendix for additional details). This approach puts greater emphasis on the lowest TDI classes.

Table 2–6. Change Agent for the Terrestrial Development Index. Classes of development and metrics, data sources, and analysis units are provided.

Change Agent	Variable class	Variable	Metric	Data sources[1]
	Transportation	Roads, railroads	Total surface area	O’Donnell and others (2014); TIGER; FRA
	Energy and minerals	Oil and gas wells, wind turbines, mines	Total surface area	State Oil & Gas Commissions, O’Donnell and Fancher (2014), FAA data
Development	Transmission structures	Communication towers, transmission lines	Total surface area	FAA data, SAGEMAP
	Agriculture[2]	Pasture, cropland	Total surface area	LANDFIRE EVT
	Urban	Urban EVT classes	Total surface area	LANDFIRE EVT

[1] See A–15 in the Appendix for additional details on data sets. TIGER = Topological Integrated Geographic Encoding and Referencing from U.S. Census data, FRA = Federal Railroad Administration; FAA = Federal Aviation Administration data, SAGEMAP = Sagebrush and Grassland Ecosystem Map Assessment Project, and LANDFIRE EVT = Landscape Fire and Resource Management Planning, Existing Vegetation Type.

[2] Regional grazing and off-highway vehicle data were not included in the index because the data were not sufficient to allow regional analyses.

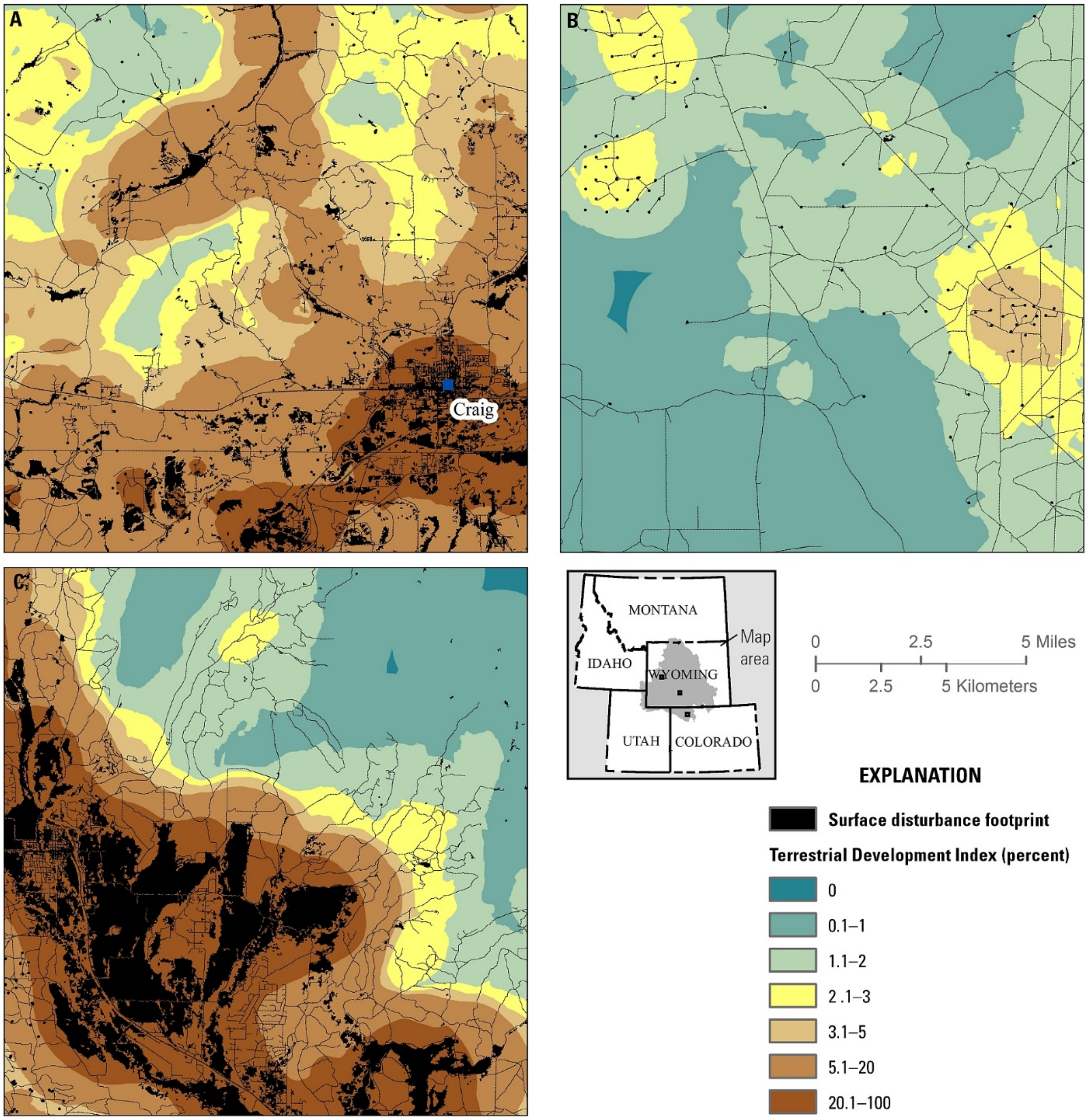

Figure 2–5. Terrestrial Development Index (TDI) score and the associated surface disturbance footprint for three landscapes in the Wyoming Basin Rapid Ecoregional Assessment project area. (*A*) A highly developed urban area near Craig, Colorado, (*B*) an oil and gas field north of Wamsutter in central Wyoming, and (*C*) an agricultural landscape near Pinedale in northwest Wyoming. The TDI scores are based on the percent of surface disturbance within a 16-square-kilometer (6.18-square-mile) moving window.

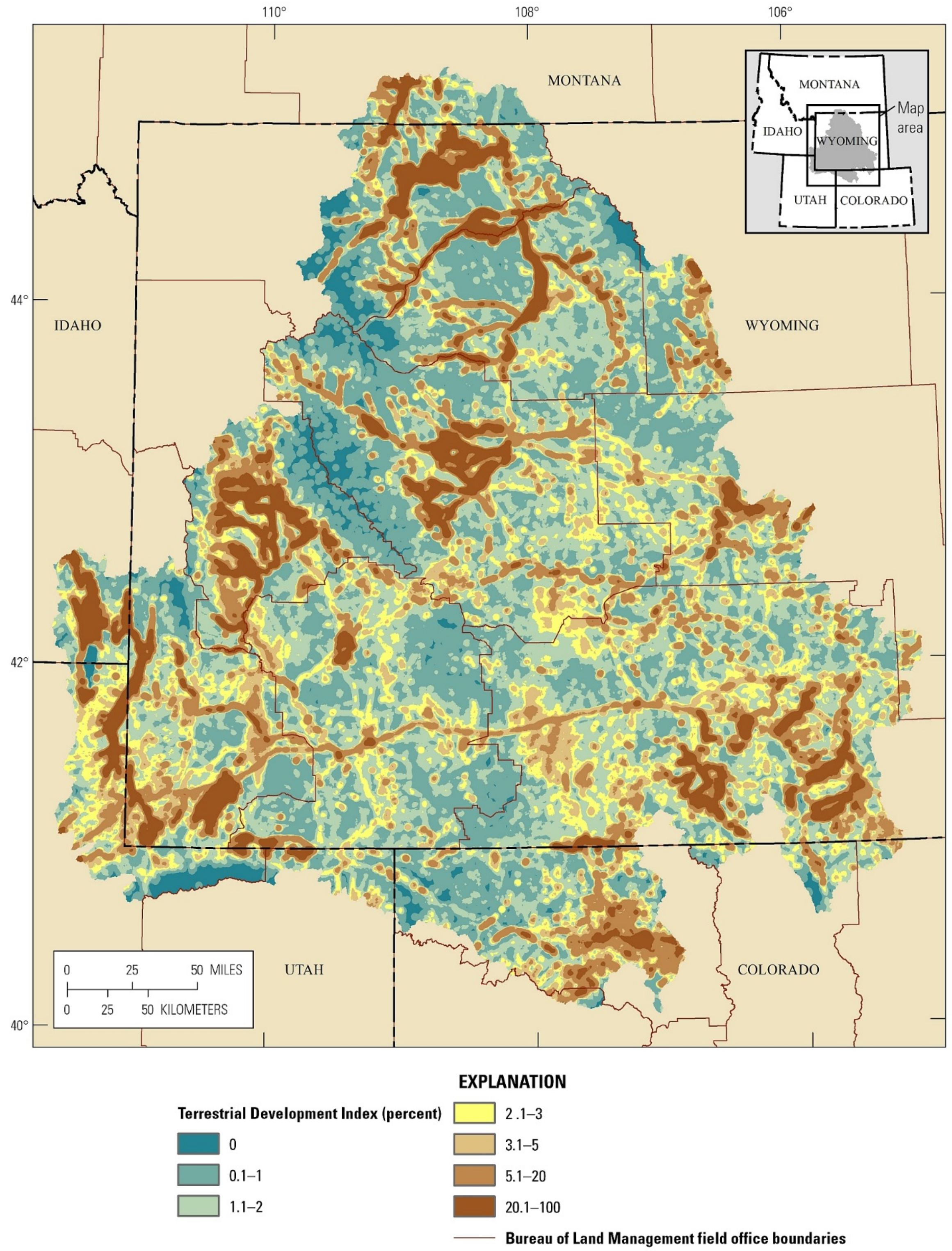

Figure 2–6. Terrestrial Development Index (TDI) for the Wyoming Basin Rapid Ecoregional Assessment project area. The TDI scores are based on the percent of surface disturbance within a 16-square-kilometer (6.18-square-mile) moving window. Relatively undeveloped areas are defined as TDI scores <1 percent.

The use of the surface area of disturbance provides a common unit for assessing a variety of development types. However, the TDI scores roughly correspond to units that are frequently used by managers. For example, road densities between 1–2 km/km^2 (1.6–3.2 mi/mi^2) corresponded to an average TDI score of 1.2 percent for roads. Likewise, well pad density of 3–6 per km^2 (8–16 per mi^2) corresponded to an average TDI score of 3.3 percent for well pads. At local scales, examination of the surface disturbance footprint in conjunction with TDI scores can also aid in the interpretation of the TDI scores (fig. 2–5).

To assess landscape intactness, we identified “relatively undeveloped areas” based on TDI ≤1 percent. Because very little area in the Wyoming Basin had TDI scores of 0 percent (fig. 2–6; see Chapter 4—Development), which is largely found within the buffer for the project area at high elevations, a threshold of TDI ≤1 percent to define relatively undeveloped areas is more practical for identifying landscape intactness and conservation potential in the Wyoming Basin. Although species differ in their sensitivity to development relating to fragmentation of habitats, we assume that large relatively undeveloped areas have the highest landscape intactness, which may increase resistance and resilience of ecological communities to drivers and stressors. Because of uncertainty in the relationship between TDI and risk from development for a particular species, we retain the entire gradient of development in the results, but we focused on the lowest and highest TDI scores to evaluate landscape-level ecological values and risks. We lack information on how species respond to the mid-range values of TDI values and the levels of development at which the transition from “intact” to “degraded” occurs, which varies among species.

Aquatic Development Index

Key ecological attributes for aquatic systems affected by development include flow regime, sedimentation regime, structural connectivity, water quality, and riparian vegetation. We calculated an overall Aquatic Development Index (ADI) using an approach similar to that used for calculating the TDI (table 2–7). The ADI is based on the synoptic human threat index developed by Annis and others (2010). The TDI was used to represent surface disturbance in the ADI. In addition, the ADI includes variables relating to water use and quality (such as dams, diversions, and road-stream crossings; table 2–7). All development variables were quantified both at the catchment level (Local ADI) and upstream contributing area for the catchment (Upstream ADI). Local-level stream segments and resulting catchments were defined using GIS based on a threshold of 3 km^2 (1.16 mi^2). All point source development variables (oil and gas pads, mines, road crossings, diversions and dams) were weighted based on distance (Annis and others, 2010). ADI scores ranged from 0 to 100 were divided into seven classes for visualization purposes based on the Wyoming Stream Integrity Index (Hargett and others, 2011). ADI scores <20 were used to represent relatively undeveloped areas (see Aquatic Development Index in the Appendix for additional details). We summarized ADI for catchments (native resolution of the data) and averaged catchment ADI scores for sixth-level watersheds (fig. 2–7).

Table 2–7. Change Agent for the Aquatic Development Index. Classes of development and metrics, data sources, and analysis units are provided.

Change Agent	Variable class	Variable	Metric	Data sources[1]
Development	Transportation	Roads, railroads	Total surface area, number of road, crossings per stream km	O'Donnell and others (2014); TIGER; FRA
	Energy & minerals	Oil and gas wells, wind turbines, mines	Number of oil and gas wells, number of wind turbines, number of mines	Oil and Gas Commissions for each State, U.S. Geological Survey wind data series, FAA data
	Water	Dams, diversions, streams under section 303D of the Clean Water Act	Number of dams, number of diversions, kilometers of stream length	State water resource data[2], Environmental Protection Agency
	Agriculture[3]	Pasture, cropland	Total surface area	LANDFIRE EVT
	Urban	Urban EVT classes	Total surface area	LANDFIRE EVT

[1] See A–15 in the Appendix for additional details on data sets. U.S. = United States; TIGER = Topological Integrated Geographic Encoding and Referencing from U.S. Census data; FRA = Federal Railroad Administration; FAA = Federal Aviation Administration data; LANDFIRE EVT = Landscape Fire and Resource Management Planning, Existing Vegetation Type

[2] Wyoming State Water Plan, Idaho Water Resources, Colorado Division of Water Resources, Montana National Resource Information System

[3] Regional grazing and off-highway vehicle data were not included in the index because the data were not sufficient to allow regional analyses.

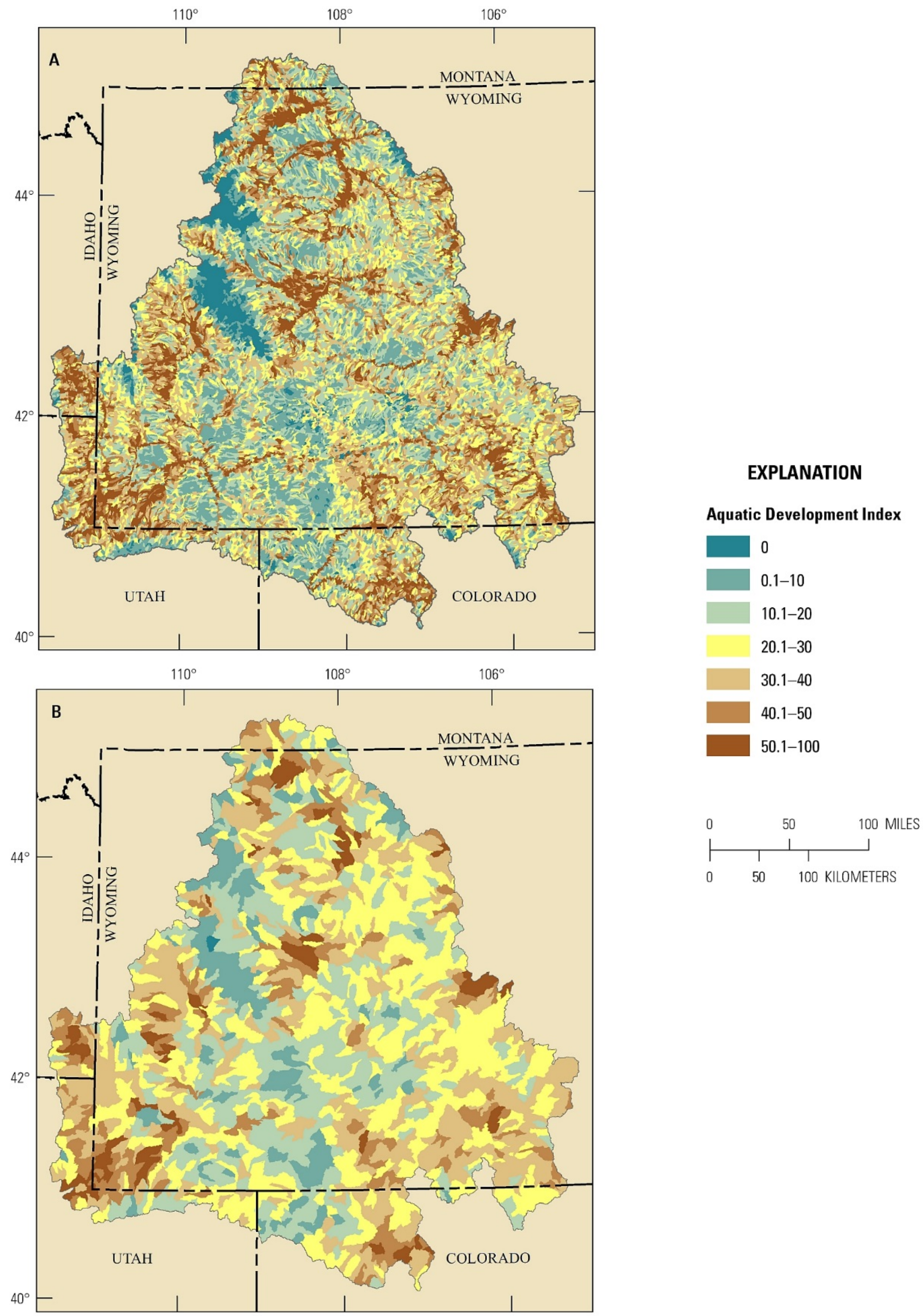

Figure 2–7. The Aquatic Development Index for the Wyoming Basin Rapid Ecoregional Assessment project area, summarized by (*A*) catchments (native resolution of dataset) and (*B*) sixth-level watersheds.

Landscape Structure: Fragmentation and Structural Connectivity

The foundation of the assessment framework uses TDI and ADI to quantify the cumulative landscape-level effects of development on individual species or assemblages, ecological communities, and for the ecoregion as a whole. The TDI and ADI quantify levels of development intensity and provide a standardized basis for comparing the potential risks from development across species and communities. To evaluate the fragmenting effects of development, we compared patch size and structural connectivity for baseline and relatively undeveloped areas for each species and community. Structural connectivity is determined by the spatial distribution of patches and patch complexes, but does not incorporate the response of species to structural connectivity. By defining levels of structural connectivity at multiple spatial scales (fig. 2−8), we assume the scales correspond to ecological processes that vary with scale, including nest-site selection and home range for local levels, dispersal and seasonal movements for landscape levels, and migration, gene flow, and range shifts at regional levels (table 2–7).

Although landscapes are clearly patchy, defining a patch depends on the species or response variable in question, as well as the scale of analysis. This is due in part to variation among species in their responses to spatial heterogeneity in the environment, and their response to spatial heterogeneity can vary across spatial and temporal scales. Likewise, functional connectivity refers to how landscape structure affects the movements of organisms and depends on a species' response to structural connectivity of its habitat (Wiens, 2002). Consequently, patch size and structural connectivity as represented here provide an index of the fragmenting effects of development across a range of scales, but the consequences for species depend on their functional response to landscape structure, which was not evaluated. There are published methods that can be used to evaluate functional connectivity (such as Compton and others, 2007; Beier and others, 2011; Cushman and others, 2013), but the short time frame of the REA was not sufficient to develop functional connectivity models for each species. Although assessing functional connectivity was beyond the scope of the REA, the results of the structural connectivity analysis and evaluation of barriers/corridors can be used to identify areas where functional connectivity analysis may be useful. Because we lack information on how species respond to the landscape structure as quantified in the REA, these results should be viewed as an index of the potential direct and indirect effects of development.

Terrestrial Landscape Structure

The effect of development on patch size was used as an index of fragmentation. Because species vary in their sensitivity to the direct and indirect effects of development, we quantified patch sizes based on distribution maps of species and communities for baseline conditions, relatively undeveloped areas (TDI ≤1 percent), and for TDI ≤3 percent, which may correspond to species with a higher tolerance for development. Baseline conditions provide a benchmark for comparisons of patch size. Knowledge of baseline conditions is necessary because some species, such as aspen or limber pine in foothill settings, naturally occur in small patches, which have important ecological functions, For example, aspen patches surrounded by arid shrublands may serve as dispersal and migration "stepping stones" for plant and animal species that require forest habitats; such stepping stones can facilitate movements across vast expanses of arid shrublands that may inhibit the movements of species closely tied to forests.

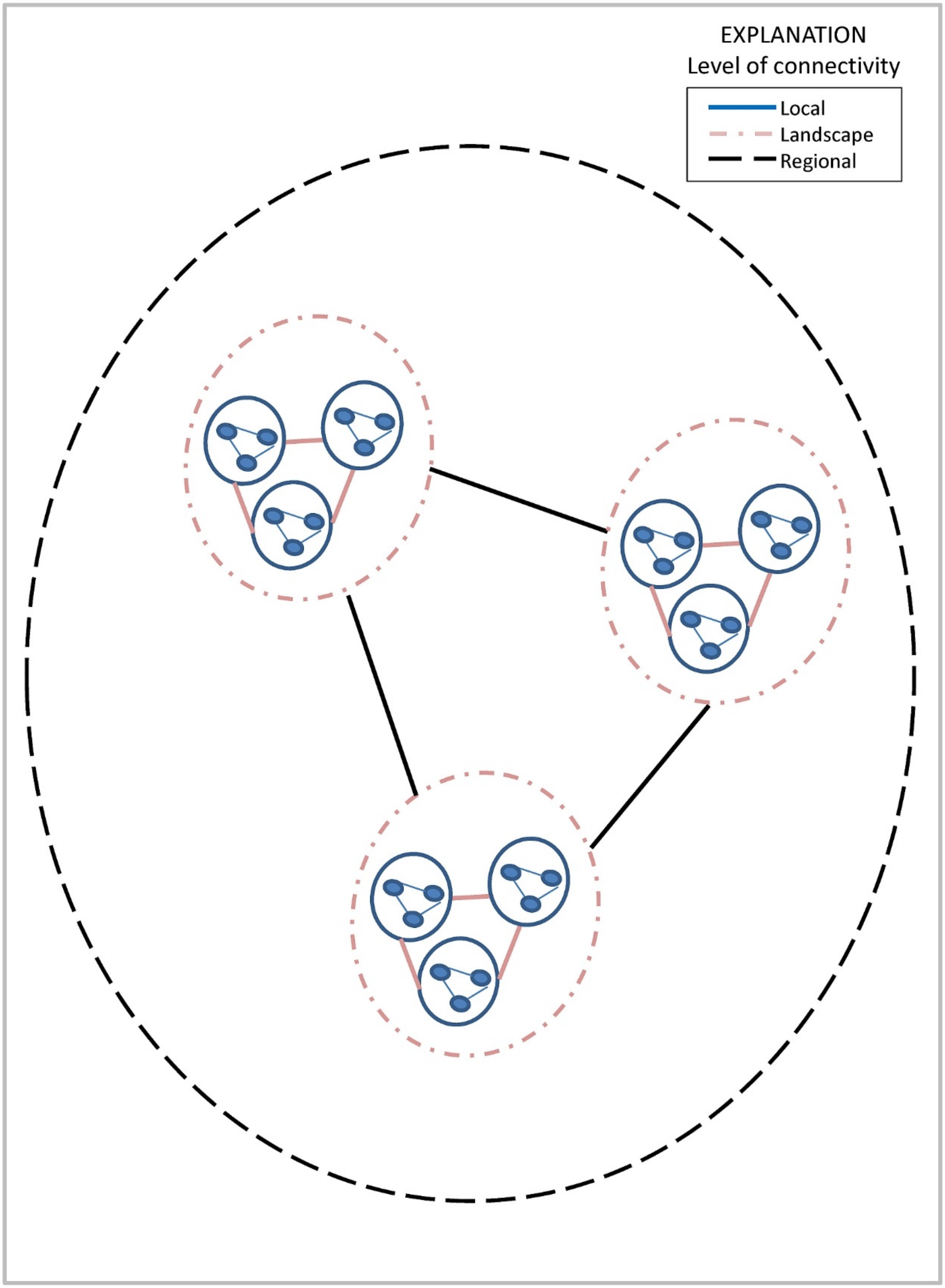

Figure 2–8. Diagram representing levels of structural connectivity for patch complexes with a nested patch structure. Patches connected at a local level are indicated by solid blue ovals and lines; landscape-level patch complexes are indicated by dashed pink ovals and solid pink lines; and regional-level patch complexes are indicated by the dashed black oval and solid black lines. Levels of connectivity correspond to discontinuities in patch connectedness at particular scales.

To evaluate structural connectivity, we first evaluated the interpatch distances at which discontinuities in patch connectedness occurred for the mapped baseline distribution of species and communities (hereafter distribution) (fig. 2–9A). Pronounced thresholds can indicate large discontinuities in the distribution of patches at particular spatial scales. Discontinuities are indicated by large increases in the maximum size of patch complexes (as a function of the total percent of the distribution). These discontinuities were used as an index of local, landscape, and regional levels of patch connectedness (figs. 2–8 and 2–9) and were used to identify the characteristic scales corresponding to each level of patch connectedness for a given species or community. For example, isolated aspen or juniper woodlands surrounded by sagebrush might exhibit a threshold at the scale corresponding to the interpatch distance separating isolated clusters of woodlands. By defining structural connectivity at multiple spatial scales, we assume these scales correspond to relevant ecological processes that may occur over a range of scales. Regional-level connectivity was based on the interpatch distance connecting >90 percent of the total distribution for a given species or community (as determined by the areas of the largest connected patch complex for a particular interpatch distance).

The graphical representation of this analysis for many species and communities showed at least 2 to 3 thresholds (see hypothetical representations in fig. 2–9), but baseline habitat for most sagebrush-associated species showed high connectivity over very short interpatch distances (fig. 2–9C). In a few cases, there were multiple small thresholds indicating heterogeneity varied rather continuously across spatial levels (fig. 2–9B). In these cases, we selected the most pronounced thresholds for identifying interpatch distances corresponding to each level of connectivity (graphical representations of connectivity analysis for each terrestrial species and community are provided in the Terrestrial Structural Connectivity Analysis section in the Appendix). The interpatch distances were used to identify and represent the scales at which structural connectivity of patches occurs, and it is a technique to simplify the complexity of multiscale spatial heterogeneity for analysis purposes.

We compared the interpatch distances of baseline conditions with relatively undeveloped areas for each species and community as an index of the potential loss of structural connectivity resulting from the fragmenting effects of development. For species that occur in more isolated patches of baseline habitat (fig. 2–9D), we mapped only baseline structural connectivity, but include interpatch distances for both baseline and relatively undeveloped areas in the summary tables for key ecological attributes and Change Agents.

We evaluated the potential for barriers and corridors between relatively undeveloped areas based on TDI scores. We assumed that areas with higher TDI scores may represent greater resistance to organism movements, whereas areas with lower TDI scores may represent potential movement corridors among relatively undeveloped areas. The degree to which they actually function as barriers or corridors depends on a species' sensitivity to the direct and indirect effects of development, the mobility of the organism, and the characteristics of the matrix (such as percent sagebrush cover) between relatively undeveloped patches. The barriers and corridors maps can be used at multiple spatial extents (such as, the entire ecoregion, district, or field office) to screen areas that may represent potential barriers and corridors. More detailed modeling at local scales (such as for a project) could provide maps that could be used for siting projects that minimize the loss of connectivity (fig. 2–10). See figures 2–11 to 2–13 for an overview of the process of evaluating key ecological attributes and effects of development on landscape structure for terrestrial species and communities.

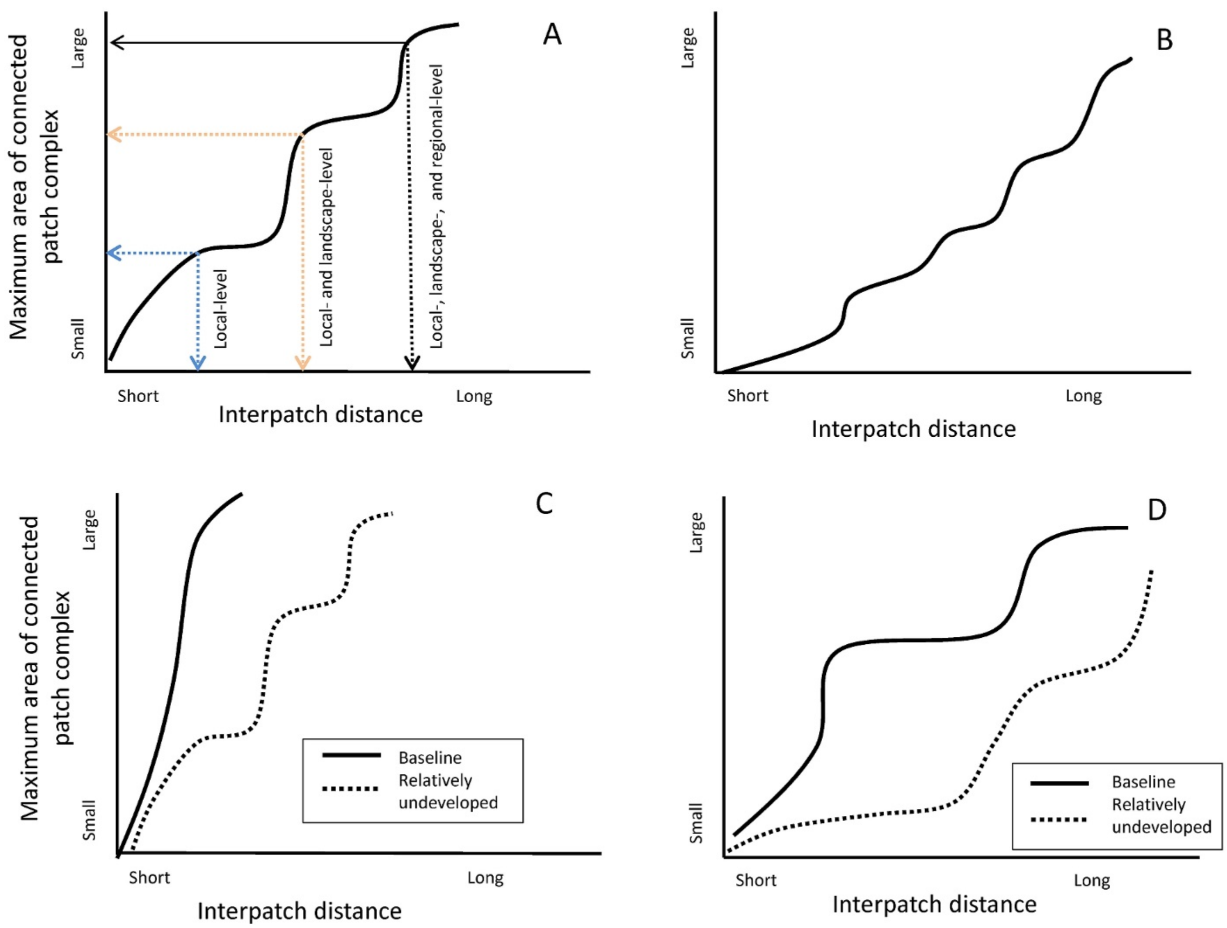

Figure 2–9. Structural connectivity is based on the relationship between the area of patch complexes and interpatch distances to define local, landscape, and regional levels of connectivity (fig. 2–8) for terrestrial ecological communities or habitats. Proximity of patches, or connectivity, is reflected by the interpatch distances. Thresholds in connectivity among patch complexes may occur at interpatch distances corresponding to the scales of patchiness in the distribution of a community or habitat. For example, the areal extent of clustered aspen patches (complexes) that are separated by a broad expanse of sagebrush may reflect connectedness thresholds corresponding to the distances separating those complexes. The thresholds are then used to determine the interpatch distances that represent local, landscape, and regional levels of structural connectivity. Regional connectivity is defined by the interpatch distance that connects >90 percent of the total area of the community or habitat. To characterize how development has affected the structural connectivity of communities, we compared interpatch distances between baseline and relatively undeveloped areas habitats at local, landscape, and regional levels. (*A*) Pronounced thresholds where connectedness changes rapidly can indicate discontinuities in the distribution of patches for particular interpatch distances, which is used to define local-, landscape-, and regional-levels of connectivity. (*B*) A lack of pronounced thresholds indicates that spatial heterogeneity varies across scales and is only loosely structured into nested levels for a specific range of interpatch distances. (*C*) Highly connected patches, characteristic of dominant ecological communities in an ecoregion, such as sagebrush steppe, can result in high regional connectivity at short interpatch distances such that no hierarchical levels are evident for baseline conditions. Fragmentation by development, however, can decrease connectivity in these systems. (*D*) Sharp discontinuities in the distribution of habitat or species (typical of aspen and juniper woodlands in the Wyoming Basin) are indicated by a large increase in patch connectedness over a small increase in interpatch distances.

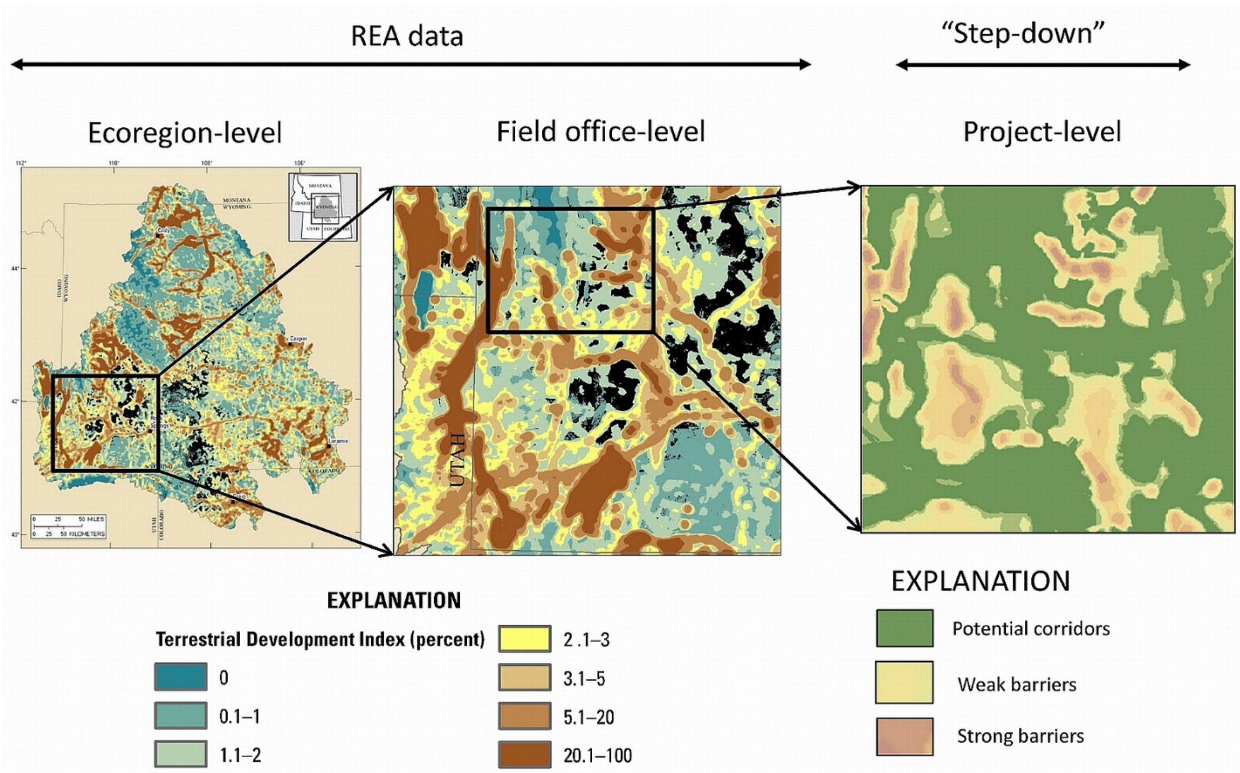

Figure 2–10. Multiple scales of information for use in the "step down" process for assessing potential barriers and corridors for pygmy rabbits. Ecoregional and field-office level analyses are based on Terrestrial Development Index (TDI) scores. High TDI scores represent potential barriers, whereas low TDI scores represent potential corridors. At a project level, a more detailed connectivity model could be developed that includes percent sagebrush cover, presence of pygmy rabbit habitat, and development levels based on the Terrestrial Development Index.

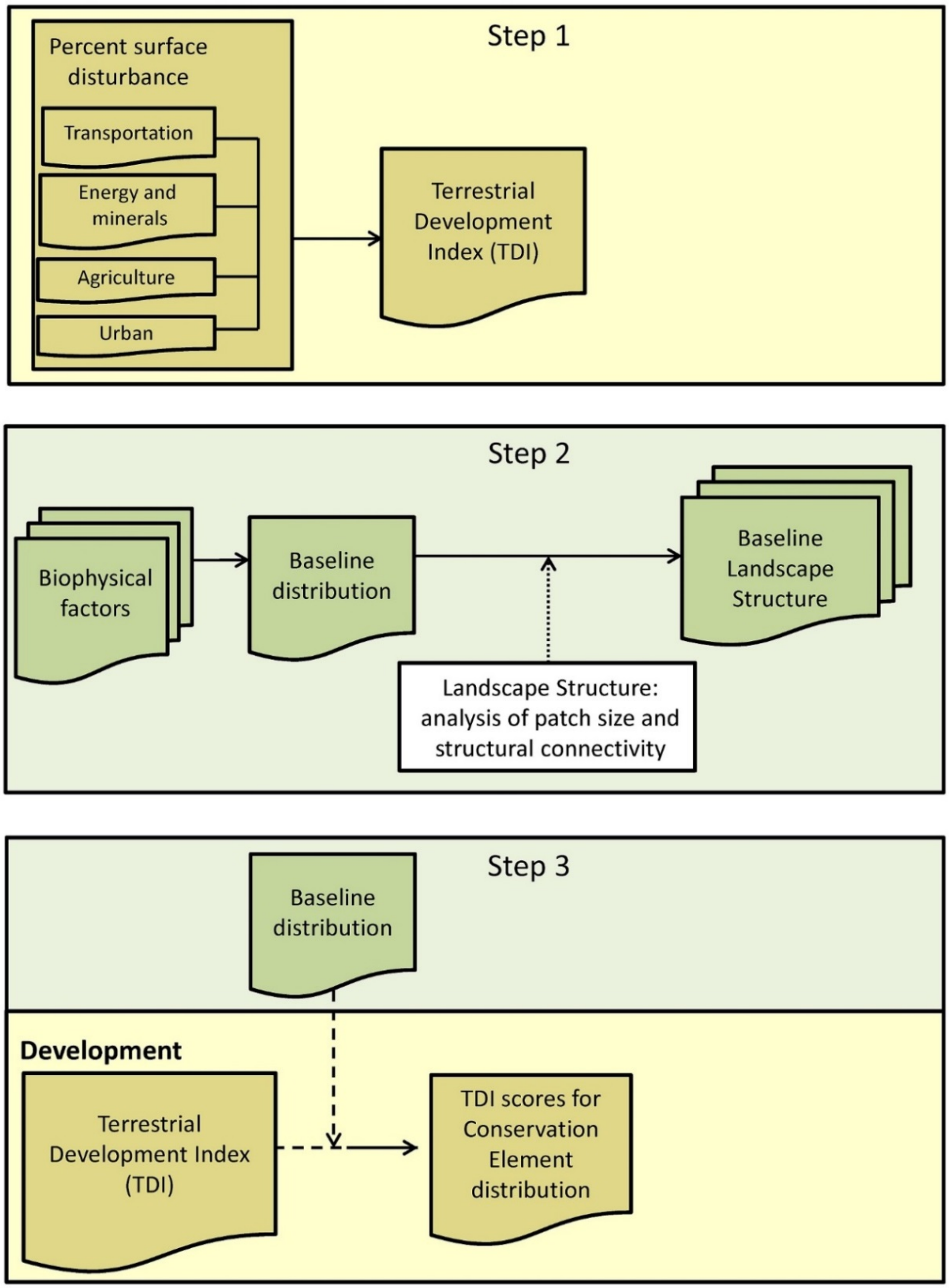

Figure 2–11. Process model (steps 1−3) in the assessment framework for terrestrial communities and species evaluated in the Wyoming Basin Rapid Ecoregional Assessment. Step 1 creates the Terrestrial Development Index (TDI). Step 2 creates baseline distribution maps and quantifies landscape structure (patch size and structural connectivity). Step 3 calculates the TDI scores for baseline distribution maps. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps. White boxes indicate analyses performed on derived maps.

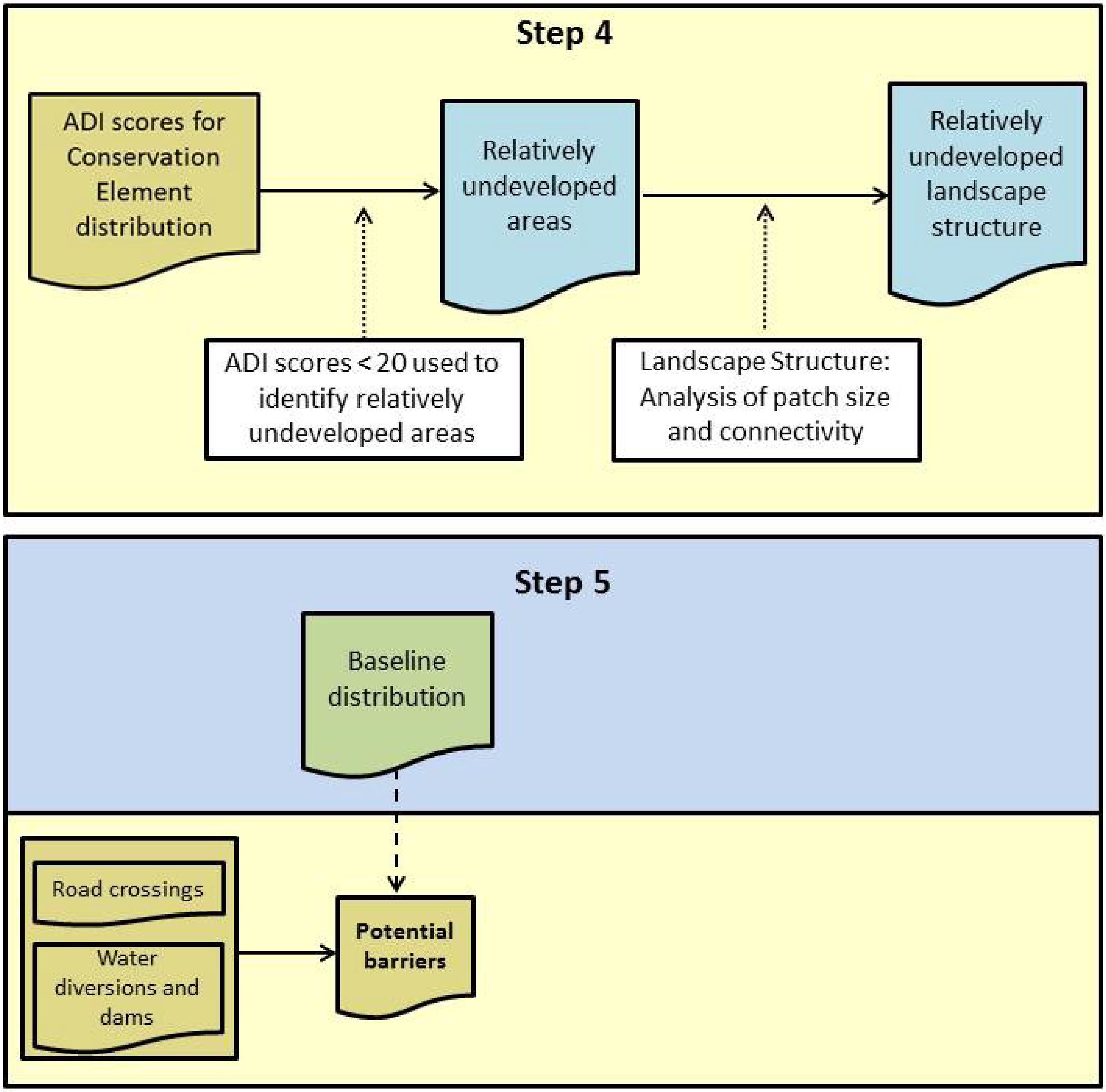

Figure 2–12. Process model (steps 4−5) in the assessment framework for terrestrial communities and species evaluated in the Wyoming Basin Rapid Ecoregional Assessment. Step 4 uses Terrestrial Development Index (TDI) scores ≤1 percent to identify relatively undeveloped areas for each distribution map and quantify landscape structure. Step 5 uses an overlay of relatively undeveloped patches on the step 1 map (ecoregion-level TDI) to identify potential barriers (high TDI scores) and corridors (low TDI scores) between relatively undeveloped patches. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps. White boxes indicate analyses performed on derived maps.

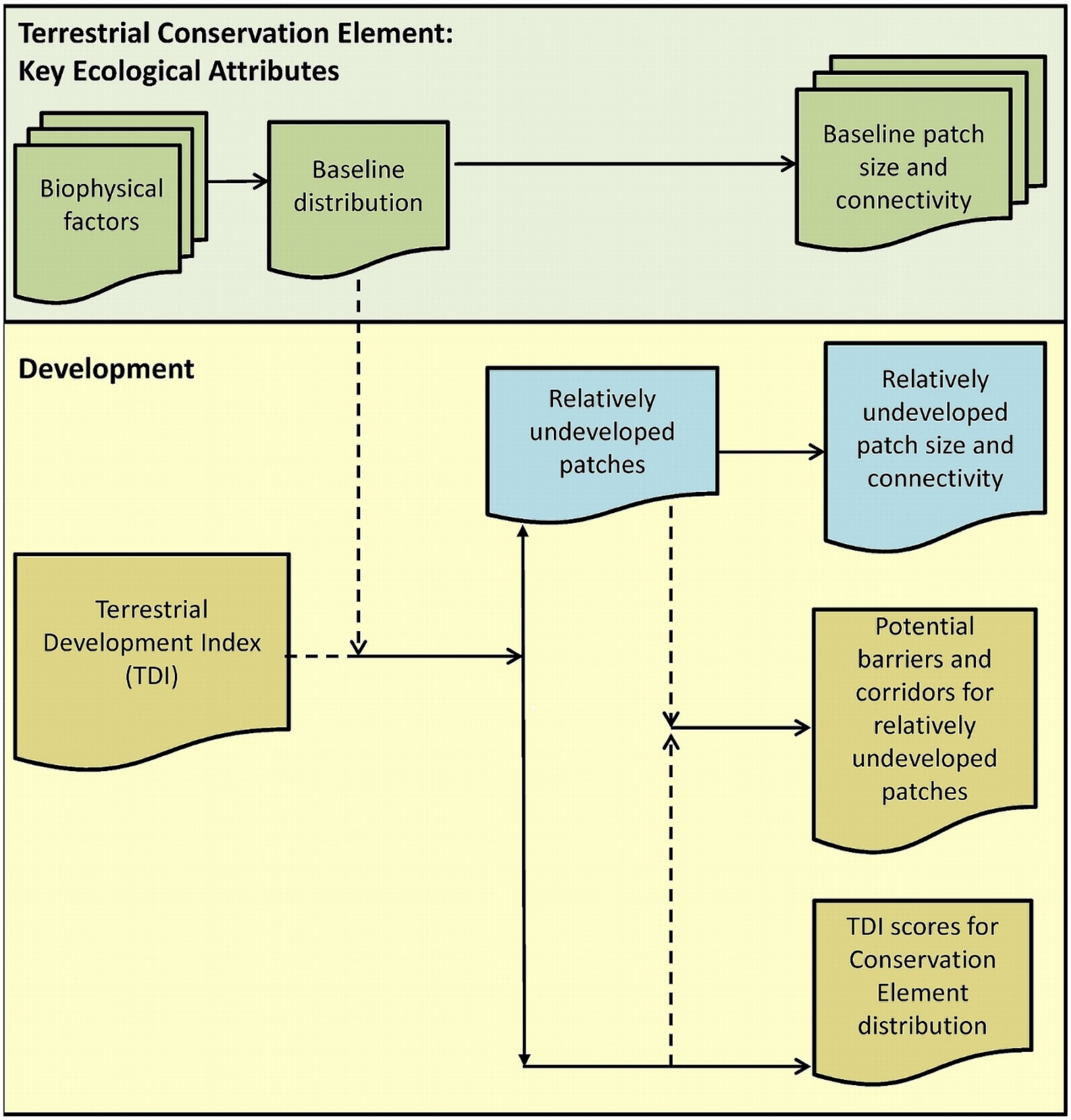

Figure 2–13. Overview of the assessment framework, which is based on the Terrestrial Development Index for terrestrial communities and species (Conservation Elements). Process models (steps 1−5) are detailed in Figures 2–11 and 2–12. The assessment framework forms the foundation of the Wyoming Basin Rapid Ecoregional Assessment. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps.

Aquatic Landscape Structure

For streams, rivers, and fish species, patch size was based on stream-segment length defined by the baseline distribution map, and for stream-segments with ADI scores <20 and ADI scores <40. The effects of development on patch size were used as an index of fragmentation. Structural connectivity was evaluated by overlaying maps summarizing the number of dams and potential barriers (diversions and stream crossings) on the distribution maps. These are described in the relevant chapters for each species, species assemblage, and for rivers and streams. Wetlands and riparian areas were handled differently, as described in the respective chapters. See figures 2–14 to 2–16 for an overview of the process of evaluating key ecological attributes and effects of development for aquatic species and communities.

Landscape Dynamics

The aspects of landscape dynamics evaluated for the REA included fire occurrence, bark beetle outbreaks, conifer-shrubland and conifer-aspen ecotone dynamics, and hydrological regime (such as mean summer flow and timing of peak flow). These are described in the key ecological attribute tables for each community or species in their respective chapters. Because generally we lack information about how human activities have altered landscape dynamics, such as disturbance regimes, that can vary over large spatial and temporal scales, we summarize available information for indicators of landscape dynamics, but do not draw strong conclusions about whether the current dynamics are consistent with historical regimes. Because of this uncertainty regarding landscape dynamics, we provide additional details in the narratives that summarize the current state of knowledge about historical dynamics and the potential consequences for communities and species.

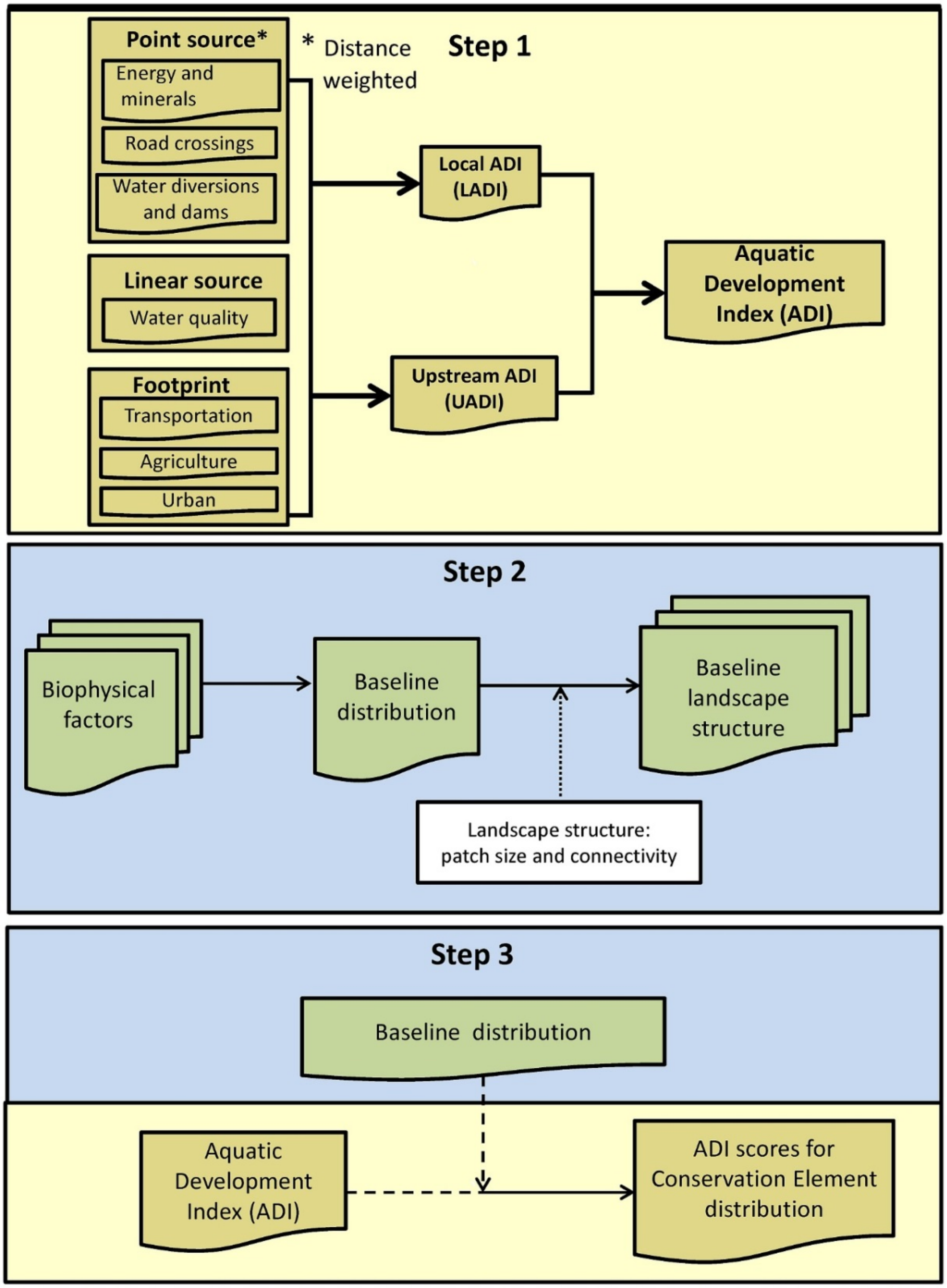

Figure 2–14. Process model (steps 1−3) in the assessment framework for aquatic communities and species evaluated in the Wyoming Basin Rapid Ecoregional Assessment. Step 1 creates the Aquatic Development Index (ADI). Step 2 creates baseline distribution maps and quantifies landscape structure (patch size and structural connectivity). Step 3 calculates the ADI scores for baseline distribution maps. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps. White boxes indicate analyses performed on derived maps.

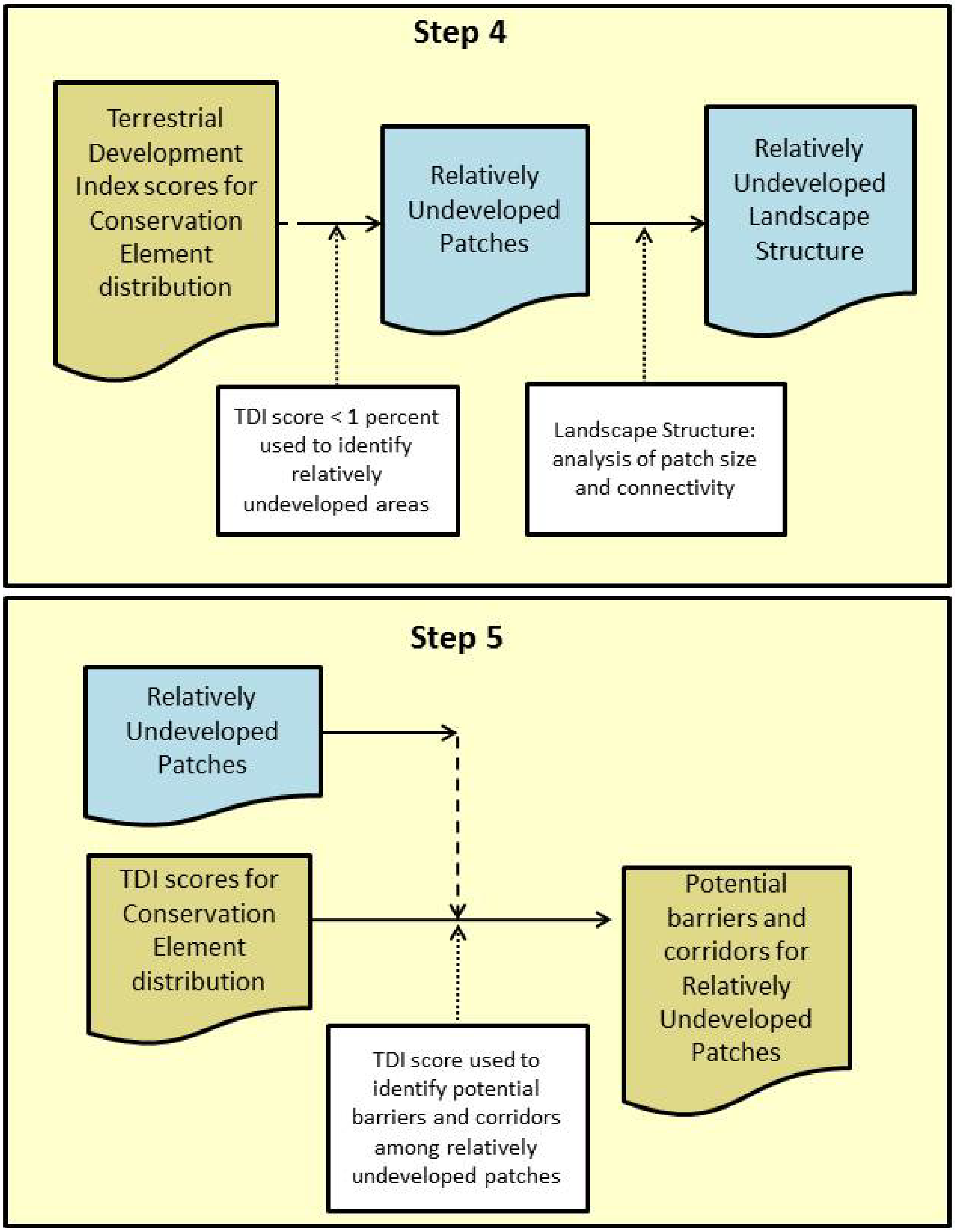

Figure 2–15. Process model (steps 4−5) in the assessment framework for aquatic communities and species evaluated in the Wyoming Basin Rapid Ecoregional Assessment. Step 4 uses the Aquatic Development Index (ADI) scores <20 for the distribution map to identify relatively undeveloped areas. Step 5 uses the number of road crossings, diversions, and dams in fifth-level watersheds as an index of potential barriers. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps. White boxes indicate analyses performed on derived maps.

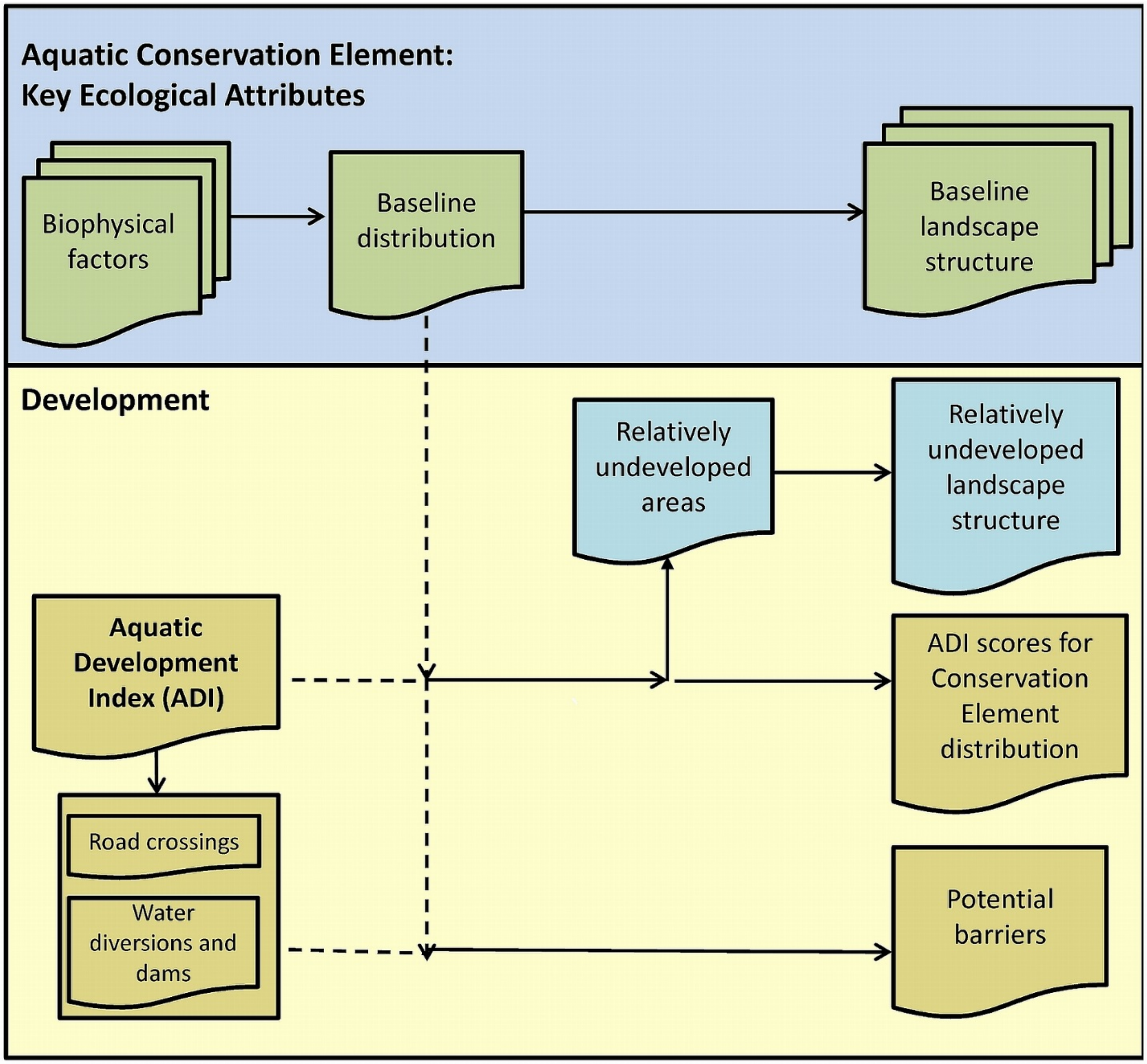

Figure 2–16. Overview of the assessment framework, which is based on the Aquatic Development Index for aquatic communities and species (Conservation Elements). Process models (steps 1−5) are detailed in figures 2–14 and 2–15. The assessment framework forms the foundation of the Wyoming Basin Rapid Ecoregional Assessment. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps.

Climate Change

Terrestrial Ecological Communities

To evaluate the potential effects of climate change on plant communities and tree species (aspen, juniper woodlands, and five-needle pines), we relied on available models of bioclimatic conditions (also called bioclimatic envelopes) suitable for ecological communities and tree species across the U.S. developed by Rehfeldt and others (2012). We developed a cross-walk between the biome classification used by Rehfeldt and others (2012) and the ecological communities based on LANDFIRE used for the REA, and applied our naming conventions for communities to this cross-walk (table 2–8 and fig. 2–17; see also A–1 in the Appendix).

Table 2–8. Crosswalk of the Wyoming Basin Rapid Ecoregional Assessment ecological communities and Rehfeldt and others (2012) biomes used for bioclimatic envelope models.

Ecological Community	Biome
Desert shrublands	Great Basin desert scrub
Sagebrush steppe	Great Basin shrub-grassland
Foothill shrublands and woodlands	Great Basin montane scrub Great Basin conifer woodlands
Montane and subalpine forests	Rocky Mountain montane conifer forest Rocky Mountain subalpine conifer forest
Plains grasslands (from LANDFIRE EVT) [1]	Plains grassland
Alpine zone (from LANDFIRE EVT) [1]	Western alpine tundra

[1] Two vegetation types originally combined with the Rapid Ecoregional Assessment ecological communities were included in climate change analysis to allow better correspondence with biomes were derived from LANDFIRE EVT = Landscape Fire and Resource Management Planning, Existing Vegetation Type.

Although there was general correspondence among the REA distributions of terrestrial communities and the bioclimatic envelope models for biomes, there were some important differences (fig. 2–17). First, Rehfeldt and others (2012) classified the sagebrush steppe and the foothill shrublands and woodlands (a community dominated by mountain big sagebrush) communities as a single biome (sagebrush shrublands), so we could not address projected climate change for these communities separately. Bioclimatic envelope models, however, were available for several tree species included in foothill shrublands and woodlands: aspen (see Chapter 15—Aspen Forests and Woodlands), limber pine, and whitebark pine (see Chapter 16—Five-Needle Pine Forests and Woodlands). In addition, the juniper woodland biome modeled by Rehfeldt and others (2012) is predominantly piñon-juniper woodlands, which currently only occurs in the southern portion of the Wyoming Basin ecoregion. Indeed, much of the expansion potential of the bioclimatic envelope for the juniper woodlands biome appears to be a consequence of projected expansion of the distribution of bioclimatic conditions suitable for piñon pine throughout the Wyoming Basin. Because of these discrepancies, the biomes represent a slightly different spatial configuration than the ecological communities (fig. 2–17). Furthermore, such fine-scale projections of the potential effects of climate change are unreliable, and broad-scale projections of climate change are more useful for providing meaningful insights into potential future shifts in the distribution of ecological communities for any one climate scenario.

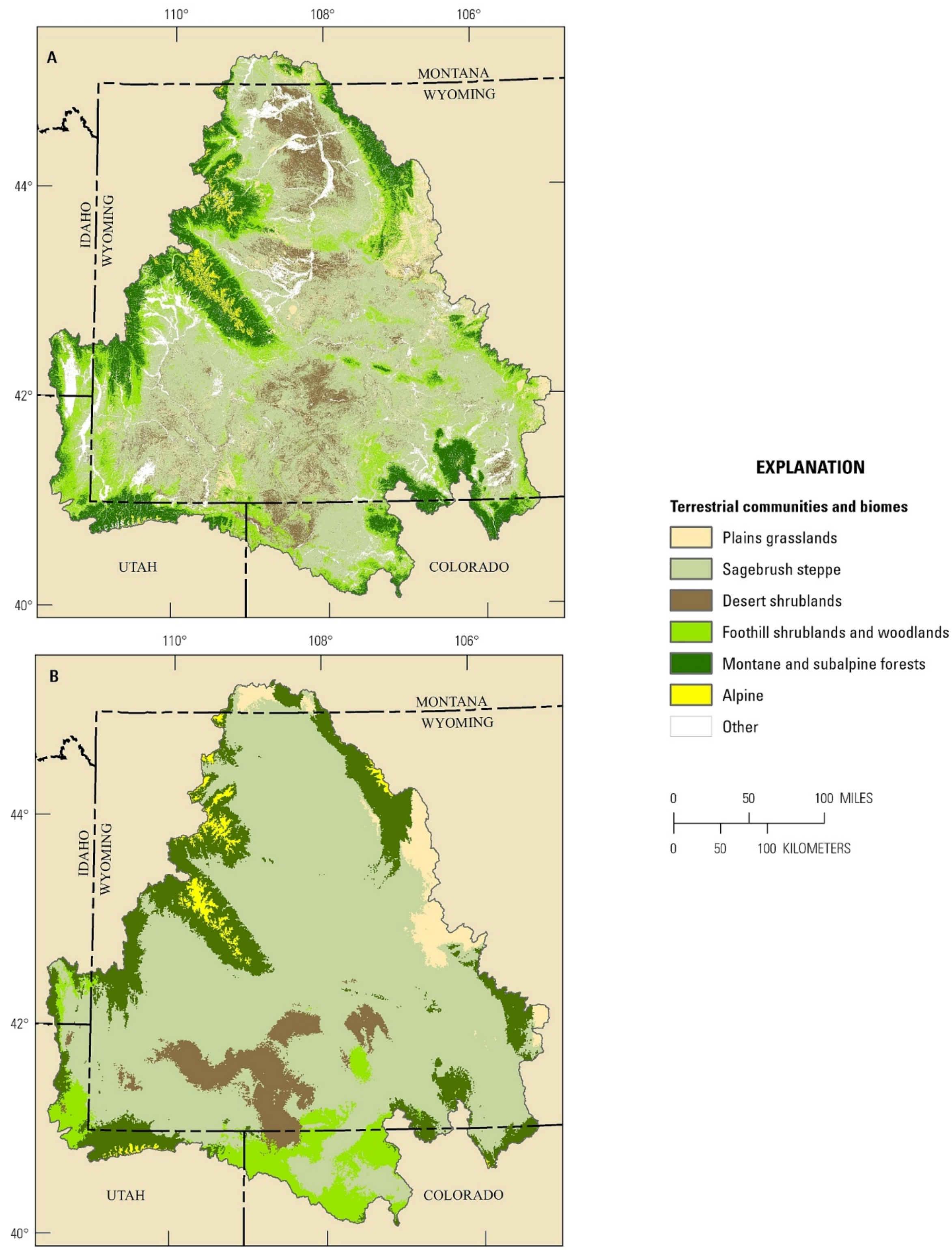

Figure 2–17. The current distribution of terrestrial communities (*A*) in the Wyoming Basin Rapid Ecoregional Assessment, as determined by LANDFIRE, and (*B*) bioclimatic conditions suitable for terrestrial biomes developed by Rehfeldt and others (2012).

To address differences between community distribution maps and bioclimatic envelopes we present the results in several ways. First, we used the community-biome crosswalk to represent the potential changes in bioclimatic envelope for biomes modeled by Rehfeldt and others (2012) for three of the climate change scenarios they evaluated (fig. 2–18). These scenarios correspond to several of the Reasonably Foreseeable Climate Scenarios addressed in the Chapter 7—Climate Analysis. For simplicity, we numbered the scenarios included in the analysis of biome shifts as climate scenario I (CCCM3), climate scenario II (GFDLCM21), and climate scenario III (HADCM3) (see Chapter 7—Climate Analysis for a description of each of these climate change models; all used emission scenario A2).

The time periods evaluated were 2030 (2016−2030), 2060 (2046−2060), and 2090 (2076−2090). The resulting maps indicate how projected climate changes could potentially affect the distribution of the bioclimatic conditions suitable for communities, as well as different potential outcomes resulting from differences among climate change model projections (fig. 2–18). The output maps only show potential for change and indicate how the bioclimatic conditions conducive for species could shift and thus, indicate potential vulnerabilities based on the climate scenarios evaluated.

The modeled bioclimatic envelope (fig. 2–17*B*) generally corresponds to the distribution of ecological communities in the Wyoming Basin (fig. 2–17*A*), but is a much broader-scale representation of areas where a species could potentially occur than the observed current distribution (as mapped by LANDFIRE). In addition to model limitations, local conditions and legacies of past disturbance, among other factors, can affect the distribution of species within otherwise suitable bioclimatic conditions. To account for the differences between the current distribution and modeled bioclimatic envelopes, we used results from climate scenario I, in 2030, and classified each modeled biome into three change categories: (1) distributions that potentially could decline because current and projected envelope distributions do not coincide, (2) distributions that are not expected to change because the current and projected envelope distributions overlap, and (3) distributions that have the potential for expansion outside the current envelope distribution. Next, we classified potential for change in the current distribution of each community or plant species by overlaying the three change categories on the baseline distribution map. See Chapter 11—Sagebrush Steppe, Chapter 12—Desert Shrublands, Chapter 14—Montane/Subalpine Forests and Alpine Zone, Chapter 15—Aspen Forests and Woodlands, Chapter 17—Juniper Woodlands, and Chapter 16—Five-Needle Pine Forests and Woodlands for results.

The differences among the potential future biome maps (fig. 2–18) illustrate the uncertainty in projecting climate and associated vegetation changes, but they provide insights into how systems might shift for the three climate scenarios. Uncertainty in model output increases for the later time periods evaluated because climate models show greater divergence through time. Despite this uncertainty, the scenarios indicate how systems could shift for particular climate scenarios and which communities and plant species have the greatest potential to decline and or expand. Based on the climate scenarios evaluated, the models indicated the potential for desert shrublands and grasslands to expand, sagebrush steppe to contract within the Basin and move northward, forests to move upslope, and alpine zones to be greatly reduced within the Wyoming Basin. These results are not predictions, but indicate the potential vulnerability of communities and species for the projected climate scenarios evaluated.

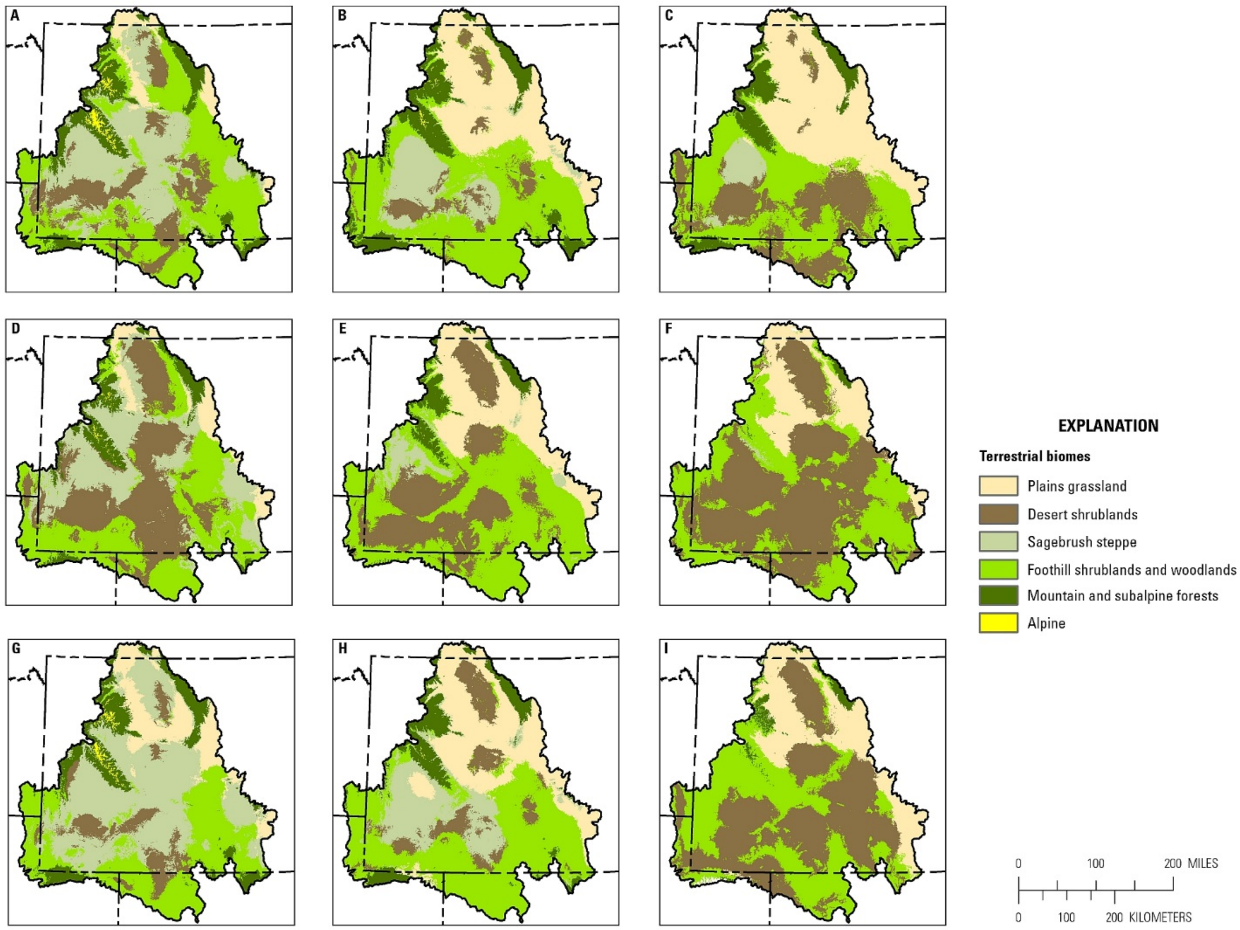

Figure 2–18. The potential distribution of bioclimatic conditions suitable for terrestrial biomes in the Wyoming Basin Rapid Ecoregional Assessment as determined by Rehfeldt and others (2012) under projected climate change scenarios. Climate scenario I (top row) for (*A*) 2030 (*B*), 2060, and (*C*) 2090; climate scenario II for (*D*) 2030, (*E*) 2060, and (*F*) 2090; and climate scenario III for (*G*) 2030, (*H*) 2060, and (*I*) 2090. Climate scenario I in 2030 (*A*) was used to evaluate the potential for changes to species and communities for the Wyoming Basin Rapid Ecoregional Assessment.

Aquatic Species and Communities

Models were available to evaluate potential effects of climate change on flow regimes for streams and rivers and associated fish habitat, and for cutthroat trout habitat (Wenger and others, 2010). Time frames evaluated by Wener and others (2010) were slightly different than time periods used for the REA (Chapter 7—Climate Analysis). The methods are described in each of the respective aquatic species and community chapters.

Invasive Species

Because of data limitations, future and projected terrestrial invasive species were addressed only for Riparian Shrublands and Forests (see also Chapter 6—Terrestrial Invasive Plant Species). For all fish and fish assemblages, we evaluated the potential for competition, predation, and (or) hybridization from introduced fish populations. Methods describing analyses are summarized in the respective chapters (also see Invasive Species section in the Appendix).

Integrated Management Questions: Landscape-Level Ecological Values and Risks

Integrated Management Questions summarize current landscape-level ecological values (based on key ecological attributes) and risks (based on Change Agents), derived from Core Management Questions. The combined ranks for landscape-level values and risks were used to rank the conservation potential of modeled distribution or mapped occurrences of species and communities. This approach summarizes information as relative ranks that can be used as a screening tool, but requires additional details included in the maps addressing Core Management Questions, in conjunction with local-level datasets that provide finer-scale details on the condition of ecological resources.

For terrestrial species and communities, conservation potential was summarized by townships based on input from the Assessment Management Team. The size of the reporting unit (93.2 km^2 [36 mi^2]) allowed us to summarize conservation potential of broad landscapes at a scale relevant to managers. Although there was variation among townships in size (mean = 79.2 km^2 [30.6 mi^2]) for the project area, a pilot analysis using a 36 mi^2 moving window indicated that variation in township size did not have a large effect on conservation potential ranks. For each terrestrial species and community, the amount of area per township was included as a landscape-level ecological value. To minimize emphasis on extremely small areas in a township in the assessment of ecological values, we put greater emphasis on large areas by establishing a minimum-area threshold for each species and community (such as 1 percent of township area; see A–19 in the Appendix for additional details on thresholds). Additional maps used to address Core Management Questions were included to assess overall ecological values if the results varied sufficiently across the project area to use as a ranking factor (such as proximity to leks for greater sage-grouse). In some cases (such as fire occurrence), available information was not sufficient to include as potential risks.

For aquatic Conservation Elements, conservation potential was summarized by fifth-level watershed based on input from the Assessment Management Team and standards for the REAs. Fifth-level watersheds in the project area average 687.3 km^2 (265.4 mi^2). For all aquatic species and communities, the amount of area per watershed was always included as a landscape-level ecological value (see A–19 in the Appendix for additional details on criteria for assigning ranks).

The TDI score was used to assess risk for terrestrial species and communities, and the ADI was used to assess risk for aquatic species and communities. The TDI/ADI ranks used were standardized and applied consistently to rank risks for each species and community (tables 2–9 and 2–10). Although species vary in their sensitivity to development (thus, the levels of development representing the highest risks varies among species), our objective was to emphasize the relatively intact areas when summarizing conservation potential, whereas areas with higher development levels (such as oil and gas fields) (figs. 2–5 and 2–6) may represent areas with restoration or development potential. This is based on the assumption that larger areas and lower development levels have higher potential for resistance and resilience of populations and communities from natural and anthropogenic disturbances. Additional Change Agents were used to assess risk for some species when available or appropriate (such as risk from sudden aspen decline, or hybridization status and risk for cutthroat trout).

Ranks for landscape-level ecological values and risks were compiled into an overall index of conservation potential for each township or watershed (fig. 2–19). The highest conservation potential represents areas that have the highest values and lowest risk. The lowest conservation potential represents areas that are ranked as having the lowest values and highest risks. Additional combinations of value and risk are possible, and were used to represent a gradient in conservation potential (fig. 2–19).

We used "lowest" and "highest" to reflect the gradient in values or risks for a given species or community (tables 2–9 and 2–10). Generally, we lack information on potential thresholds at which the value of a township or watershed may represent lower value or higher risks to a species or community. In a few cases (such as greater sage-grouse), published information on values for different areas of habitat evaluated at an appropriate spatial scale (similar to the reporting unit we used), was available. In most cases, however, such information was lacking and consequently we established criteria for assigning ranks (such as equal subsets of the data based on area by township, or biologically meaningful breakpoint for highly skewed data; see A–19 in the Appendix for additional details on the criteria) based on statistical properties of each variable.

Because size of areas or patches does not always indicate higher value, local-level information is important for evaluating conservation potential for particular areas. Furthermore, the ranks for values and risks, and conservation potential are relative (not absolute) and vary among species (for example, depend on differences among species in sensitivity to disturbance or area effects). Consequently, the lowest rank for one species may not be directly comparable to the lowest rank for another species if they vary in sensitivity to development.

Table 2–9. Example of the used to summarize landscape-level ecological values and risks for terrestrial Conservation Elements. Ranks were combined into an index of conservation potential for the Wyoming Basin Rapid Ecoregional Assessment.[1]

		Relative Rank			
	Variables[1]	Lowest	Medium	Highest	Description
Values	Area	0–35	35–79	>79	Percent of township classified as sagebrush steppe
Risks	Terrestrial Development Index (TDI)	0–1	1–3	>3	Mean TDI by township

[1] Townships were used as an analysis unit/reporting unit for conservation potential based on input from the Bureau of Land Management. A minimum threshold based on total area per township was established for each species or community to minimize the emphasis on extremely small areas. Break points for ranks of area for each community and species were derived from equal subsets of the data such that the number of townships in each rank is approximately the same. TDI ranks were consistently applied to all communities and species (see A–19 in the Appendix for details on criteria for assigning ranks for values and risks and threshold levels)

Table 2–10. Example of the used to summarize landscape-level ecological values and risks for aquatic Conservation Elements. Ranks were combined into an index of conservation potential for the Wyoming Basin Rapid Ecoregional Assessment.[1]

	Variables	Relative Rank: Lowest	Relative Rank: Medium	Relative Rank: Highest	Description
Values	Perennial stream density	<0.13	0.130.34	>0.34	The ratio of perennial stream length to the area of fifth-level watershed
	Ephemeral/ intermittent stream density	<0.12	0.12–1.56	>1.56	The ratio of ephemeral/intermittent stream length to the area of fifth-level watershed
Risks	Aquatic Development Index (ADI)	0–20	20–40	>40	Mean ADI
	Number of dams	0	1–2	>2	Number of dams

[1] Fifth-level watersheds were used as an analysis unit/reporting unit for conservation potential based on input from Bureau of Land Management. Break points for ranks of area for each community and species were derived from equal subsets of the data such that the number of townships in each rank is approximately the same. ADI ranks were consistently applied to all communities and species (see A–19 in the Appendix for additional details on criteria for assigning ranks for values and risks.)

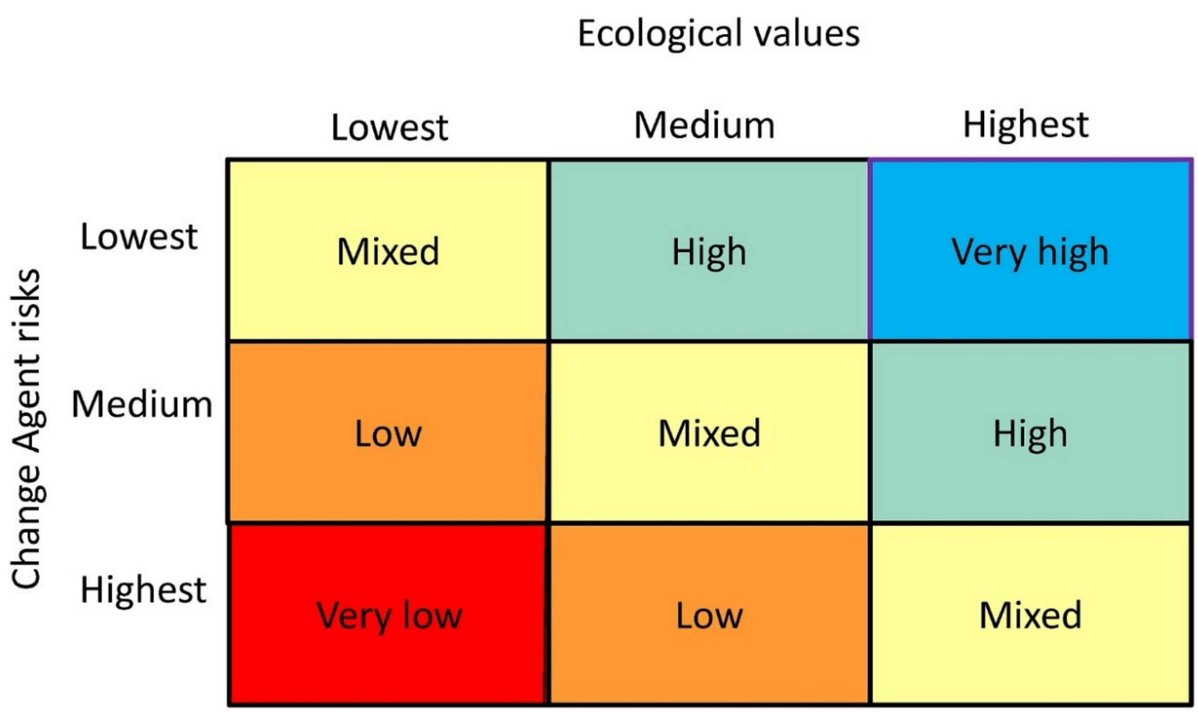

Figure 2–19. Relative ranks of conservation potential based on relative ranks of landscape-level ecological values and risks as represented by Change Agents. Very high conservation potential represents areas that have the highest value and lowest risk for a Conservation Element (blue). The lowest conservation potential represents areas that are ranked as having the lowest landscape-value and highest risks (red). Other possible combinations of ranks are indicated and represented by different colors.

Overall Process Model for Conservation Element Assessments

The process model used to conduct assessments of the Conservation Elements is summarized in table 2–20.

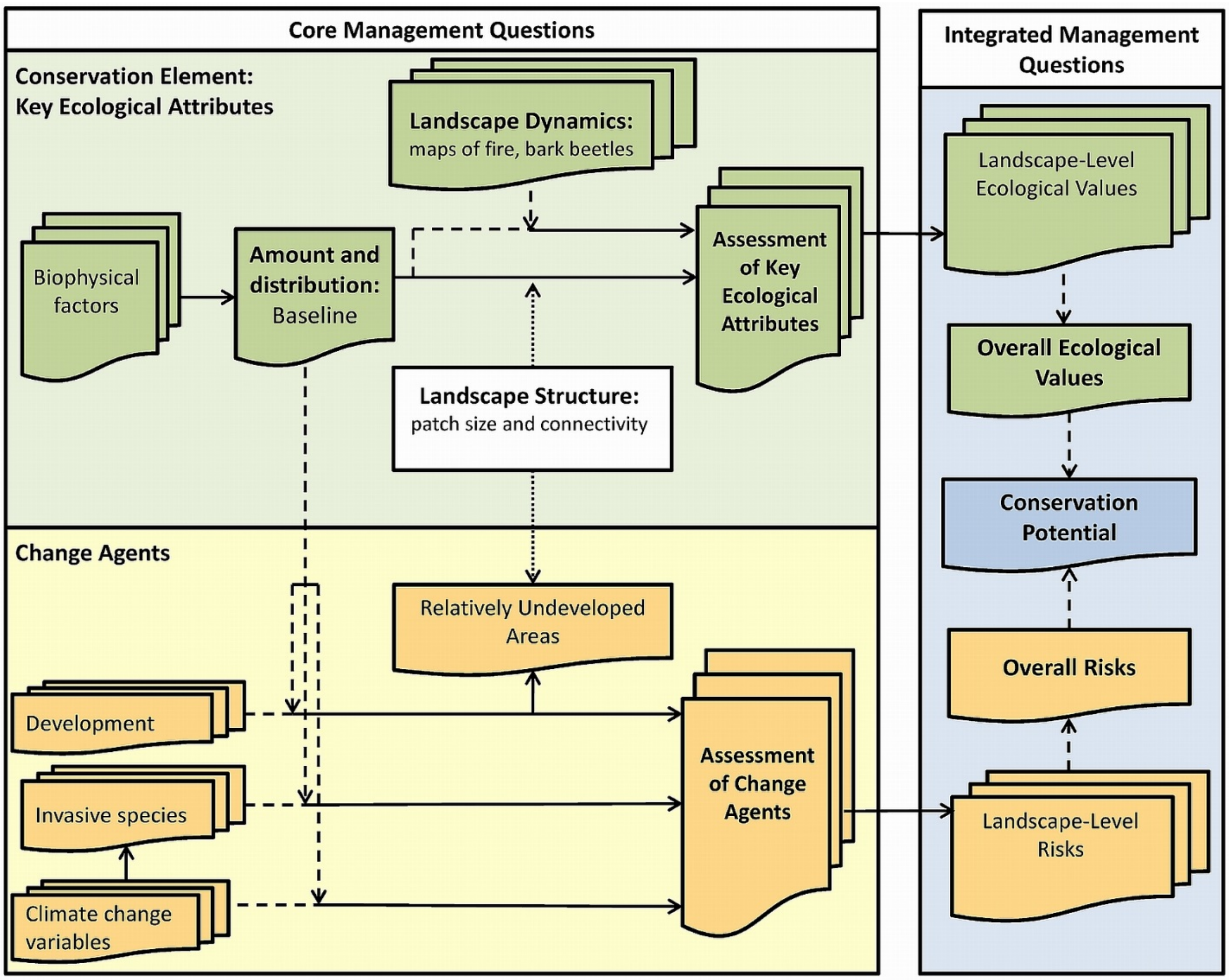

Figure 2–20. Overview of the process used to address the Core and Integrated Management Questions for each Conservation Element. Polygons indicate input and output maps used to address Core and Integrated Management Questions. Solid lines indicate the source of maps used for derived maps, dashed lines indicate map overlays are used to derive maps, and dotted lines indicate analyses performed on maps. White boxes indicate analyses performed on derived maps.

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