

2013

Integrating biodiversity and drinking water protection goals through geographic analysis


James D. Wickham

U.S. Environmental Protection Agency, wickham.james@epa.gov

Curtis H. Flather

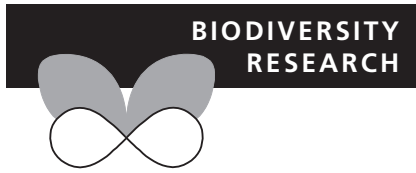
USDA Forest Service

Follow this and additional works at: <http://digitalcommons.unl.edu/usdafsfacpub>

 Part of the [Forest Biology Commons](#), [Forest Management Commons](#), [Other Forestry and Forest Sciences Commons](#), and the [Plant Sciences Commons](#)

Wickham, James D. and Flather, Curtis H., "Integrating biodiversity and drinking water protection goals through geographic analysis" (2013). *USDA Forest Service / UNL Faculty Publications*. 332.
<http://digitalcommons.unl.edu/usdafsfacpub/332>

This Article is brought to you for free and open access by the U.S. Department of Agriculture: Forest Service -- National Agroforestry Center at DigitalCommons@University of Nebraska - Lincoln. It has been accepted for inclusion in USDA Forest Service / UNL Faculty Publications by an authorized administrator of DigitalCommons@University of Nebraska - Lincoln.



Integrating biodiversity and drinking water protection goals through geographic analysis

James D. Wickham^{1*} and Curtis H. Flather²

¹National Exposure Research Laboratory, Office of Research and Development, US Environmental Protection Agency, Research Triangle Park, NC 27711, USA, ²USDA, Forest Service, Rocky Mountain Research Station, Fort Collins, CO 80526, USA

ABSTRACT

Aim Biodiversity and drinking water share a common interest in land conservation. Our objective was to identify where that common interest occurs geographically to inform conservation planning.

Location The study focused on 2112 eight-digit hydrologic units (watersheds) occurring in the conterminous United States.

Methods Data on aquatic-dependent species occurrence, drinking water intakes, protected land status and land cover change were compiled for each watershed. We compared these four datasets after defining ‘hotspots’ based on attribute-specific thresholds that included (1) the 90th percentile of at-risk aquatic biodiversity, (2) with and without drinking water intakes, (3) above and below the median percentage of protected land and (4) increase in urban land above and below a 1% threshold between 2001 and 2006. Geographic intersections were used to address a number of questions relevant to conservation planning including the following: What watersheds important to aquatic biodiversity are also important to drinking water? Which watersheds with a shared stake in biodiversity and drinking water protection have inadequate land protection? Which watersheds with potentially inadequate amounts of protected lands are also undergoing relatively rapid urbanization?

Results Over 60% of the watersheds that were determined to be aquatic biodiversity hotspots also had drinking water intakes, and approximately 50% of these watersheds had less than the United States median amount of protected land. A total of seven watersheds were found to have shared aquatic biodiversity/drinking water values, relatively low proportions of protected lands and a relatively high rate of urbanization. The majority of these watershed occurred in the south-eastern United States, with secondary occurrences in California.

Main conclusions Geographic analysis of multiple ecosystem services can identify areas of shared land conservation interest. Locations where ecosystem commodities and species conservation overlap has the potential to increase stakeholder buy-in and leverage scarce resources to conserve land that, in this case study, protects both biodiversity and drinking water.

Keywords

At-risk species, conservation planning, ecosystem services, geographic information system, protected lands, urbanization.

*Correspondence: James D. Wickham, National Exposure Laboratory, U.S. EPA Office of Research and Development, MD: 243-05, Research Triangle Park, NC 27711, USA.
E-mail: wickham.james@epa.gov

INTRODUCTION

Setting aside land from future development is a long-standing means of biodiversity protection (DellaSala *et al.*, 2001; Possingham *et al.*, 2006). Although other means are

recognized (Bean, 2000), removal of land from future development through outright acquisition, easements, rent or other mechanisms is recognized as a critical element for conservation of biodiversity (Shaffer *et al.*, 2002; Phalan *et al.*, 2011). Incorporation of geography provided a needed boost

to locating areas to set aside for biodiversity conservation (Scott *et al.*, 1987; Flather *et al.*, 1998, 2009). Development of geographic databases on the occurrence of numerous taxa provided synoptic views of species concentrations (Flather *et al.*, 1998; Shriner *et al.*, 2006) that could then be compared with existing information on status of land protection to guide selection of land areas to set aside from future development.

Identification of candidate conservation areas is central to biodiversity protection. Conservation planning software such as MARXAN was developed to identify the fewest number of (or most cost efficient) conservation areas that protected the greatest number of species (Ball *et al.*, 2009; Watts *et al.*, 2009). Hub and corridor models are inherently geographic approaches focused on identification of areas for conservation that should serve as refugia or connectors between refugia (Benedict & McMahon, 2006; Beier *et al.*, 2008; Wickham *et al.*, 2010). Population viability models integrate survival and fecundity with geographic data on habitat which can then be linked with conservation planning models to refine reserve selection (Carroll *et al.*, 2003; Beissinger *et al.*, 2009). Graph theory can be used to characterize an area as a set of habitat and linkages between them, which can be used to identify areas for conservation (Minor & Urban, 2008).

More recently, land conservation has been recognized as an important element of drinking water protection (Wickham *et al.*, 2011). One of the more well-known examples of a public policy decision based on valuing clean drinking water as an ecosystem service involved New York City's decision in 1996 to protect and restore land in the Catskill/Delaware watershed rather than investing in a water filtration system (National Research Council, 2005). During this same period, amendments to the Safe Drinking Water Act (SDWA) shifted its emphasis from contaminant detection to source water protection (US Environmental Protection Agency, 1997). In the context of drinking water, source water refers to the water entering a drinking water facility, which, in essence, is the runoff from the watershed upstream of a drinking water intake. The shift in emphasis to source water protection recognized that the quality of untreated water entering a drinking water treatment plant directly related to the cost of treating the water and the risk of waterborne pathogens and pollutants entering the drinking water supply. SDWA's new focus on source water protection is consistent with several studies that report the cost and benefits of land conservation for drinking water protection (National Research Council, 2000; Dudley & Stolton, 2003; Ernst *et al.*, 2004; Mehaffey *et al.*, 2005; Postel & Thompson, 2005).

With the shift in SDWA emphasis to source water, land use management is now an important element of two areas of study that are not usually regarded as overlapping or sharing common interests – biodiversity conservation and drinking water protection (Wickham *et al.*, 2011). The objective of this research is to demonstrate opportunities to align biodiversity and drinking water protection goals. The demonstration is conducted using spatial analysis of geographic data on species

occurrence and drinking water intake locations. Alignment is demonstrated by identifying places where land conservation will likely benefit protection of both biodiversity and drinking water.

METHODS

Data

Data on aquatic-dependent species distributions, drinking water intake locations, status of land protection and land cover (including land cover change) were used for the analysis. The US Geological Survey (USGS) eight-digit hydrologic units (watersheds) were used as the analysis unit. There are 2112 watersheds for the conterminous US in the USGS eight-digit hydrologic unit data.

Aquatic-dependent species occurrence by watershed was compiled from NatureServe's central databases (McNees, 2010) in support of the US Forest Service 2010 Resource Planning Act (RPA) assessment (see Loftus & Flather, 2012). Aquatic dependence was defined as relying on marine, estuarine, lacustrine, palustrine or riverine habitats for a significant part of the species' life cycle. Species groups included vascular plants, birds, mammals, reptiles, amphibians, non-vascular plants, fungi and lichens, invertebrates and freshwater fish. Marine species were not included. The final dataset included tallies of the total number of aquatic-dependent species and counts by conservation status rank (Table 1) for each watershed. There were no data on species occurrence for 51 of the 2112 eight-digit hydrologic units within the conterminous United States, leaving a total of 2061 used in our analyses. Many of the 51 watersheds without species data overlapped the Canadian border.

Drinking water intake locations (x, y) were provided by the US EPA Office of Ground Water and Drinking Water. The dataset included a total of 5265 intake point locations (see Wickham *et al.*, 2011). The point locations were for surface water sources only. Ground water sources were not included.

Land cover composition and land cover change were compiled by watershed from NLCD 2006 (Fry *et al.*, 2011). The NLCD 2006 database includes land cover for ca. 2001 and ca. 2006 based on temporal mapping to identify change. The difference in the amount of urban land between ca. 2001 and ca. 2006 was used as an estimate of urbanization. None of the watersheds lost urban land over the approximate 5-year

Table 1 NatureServe conservation status ranks

Rank	Description
G1	Critically imperilled
G2	Imperilled
G3	Vulnerable to extirpation or extinction
G4	Apparently secure
G5	Demonstrably widespread, abundant and secure

source: <http://www.natureserve.org/explorer/ranking.htm>.

period. Urban was defined as the sum of the four urban classes in the NLCD data; NLCD land cover class definitions are available at http://www.mrlc.gov/nlcd06_leg.php.

Land protection status was derived from the Protected Areas Database (DellaSala *et al.*, 2001), version 1.2 (<http://gapanalysis.usgs.gov/padus/data/download>). All lands in GAP categories 1, 2 and 3 as defined by Scott *et al.* (1993) were designated as protected (Table 2).

Analysis

The total number of species occurring in an area was expected to increase as some fractional power of area (Rosenzweig *et al.*, 2011). Based on linear regression analysis (SAS 9.3; SAS Institute, Cary NC, USA), we found no evidence that aquatic-dependent species counts within a watershed varied as a linear ($F = 1.13$; $P > 0.29$) or power function (data log–log transformed; $F = 0.01$; $P > 0.93$) of watershed area. Our failure to find an area relationship was consistent with Munepeeraikul *et al.* (2008) who found that freshwater fish species richness was explained not by watershed area, but by drainage area (the total area drained by a watershed which includes the area of the focal eight-digit hydrologic unit and all the units upstream from it), and the average annual runoff production as an indicator of resources available for fish production. Pearson product–moment correlation did reveal that the number of at-risk aquatic species was positively related to the total aquatic species richness ($r = 0.64$; $P < 0.0001$). This was expected as all other things being equal (e.g. the species abundance distribution), watersheds with larger species pools would be expected to have a greater number of rare (and therefore more likely to be classified as at-risk) species. To control for the species’ pool size effect, we focused our analysis on the proportion of at-risk species (No. of at-risk/total No. of species) within each watershed in our geographic comparisons with protected land and urban growth. Analyses using

proportions of at-risk species are reported in the main text, and analyses based on simple count data (for both total aquatic-dependent species and at-risk species) are reported as Supplemental Material.

Spatial analysis was based on intersecting the four datasets within a geographic information system. Thresholds were used to guide the intersections. The 90th percentile was used as a threshold to define watershed hotspots of biodiversity. The 90th percentile was applied to the proportion of at-risk species, as well as total number of species in the watershed and the number of at-risk species in the watershed. Watersheds were considered hotspots when the 90th percentile was equalled or exceeded. The 90th percentile values were 0.05066 (proportion of at-risk species), 20 (number of at-risk species) and 473 (total species richness). At-risk was defined as the number of species in categories G1, G2 and G3 (Table 1; as in Robles *et al.*, 2008). The number of watersheds included in the 90th percentile was not consistently 10% of the 2061 watersheds used in the analyses because of ties. The threshold used for number of drinking water intakes was 1, that is, at least one drinking water intake occurred in the watershed. The threshold used for percentage of protected land within a hydrologic unit was 9% – the median value of percentage of protected land across the 2061 watersheds with species distribution data. The threshold used for percentage change in urban land was 1%. Change in percentage urban was calculated as the percentage of the area of the hydrologic unit that was urban in 2006 minus the percentage of the area of the hydrologic unit that was urban in 2001. A low land cover change percentage threshold was used because of the overall large size of the eight-digit hydrologic units. The median area of the eight-digit hydrologic units is greater than 326,700 ha, which translates into a gain of urban land of more than 3260 ha (32.6 km²) for a 1% threshold. The geographic intersections using the thresholds can be used to organize the watersheds into several different categories (Table 3), such as watersheds that have high number of

Table 2 Land protection categories (from Scott *et al.*, 1993; see also DellaSala *et al.*, 2001)

Category	Description
1	There is an active management plan designed to maintain natural state; natural disturbance events proceed without interference or are mimicked (e.g. Nature Conservancy Preserves).
2	Managed for natural values, but allowed uses of the land may degrade quality (e.g. some natural wildlife refuges).
3	Most non-designated public lands; legal mandate prevents conversion to anthropogenic use with some exception; protection of United States federally listed endangered, threatened or candidate species (e.g. US Forest Service lands).
4	Private land and public land without agreements to maintain native species and natural communities (e.g. urban and agricultural areas).

Table 3 Description of hotspot watersheds (WS) categorized by number of drinking water intakes, percentage of protected land and percentage increase in urban land

Category	For WS ≥ 20 at-risk species or ≥ 437 total species		
	No. of Drinking water intakes	% Protected land	% Increase in urban land
1	0	> 9	< 1
2	0	> 9	≥ 1
3	0	≤ 9	< 1
4	0	≤ 9	≥ 1
5	≥ 1	> 9	< 1
6	≥ 1	> 9	≥ 1
7	≥ 1	≤ 9	< 1
8	≥ 1	≤ 9	≥ 1

Thresholds are ≥ 20 (No. at-risk species), ≥ 437 (species richness), ≥ 1 (number of drinking water intakes), $\leq 9\%$ (percentage of protected land) and $\geq 1\%$ (increase in urban land).

species, a low percentage of protected land, a high rate of urbanization and include at least one drinking water intake.

There was no species occurrence data for non-vascular plants, fungi and lichens, invertebrates and freshwater fish for 473 watersheds concentrated in the central United States (see Fig. S1 in Supporting Information). The effect of the missing data was assessed by comparing watersheds in the 90th percentile with and without occurrence data for these species groups.

RESULTS

Drinking water intake locations split watersheds that are biodiversity hotspots into groups with and without a collateral stake in drinking water protection. For the proportion of at-risk species, 67% (139 of 207) of the hotspot watersheds had a drinking water intake (Fig. 1). For the simpler hotspot measures of at-risk species counts and total species richness counts, the proportions of the hotspot watersheds with a drinking water intake were 67% and 57%, respectively (see Figs S2 and S3 in Supporting Information). More than half of the watersheds identified as species hotspots have a collateral stake in drinking water protection.

Inclusion of percentage of protected land provided further thematic resolution to the dichotomous grouping of species hotspot watersheds into those with and without drinking water intakes by subdividing the two groups into those with

and without an ‘adequate’ amount of protected land. For the proportion of at-risk species, there were $2.3 \times$ more watersheds with at least one drinking water intake and less than the median amount of conserved land than watersheds without a drinking water intake and less than the median amount of conserved land (Fig. 1).

Including urbanization along with number of drinking water intakes and percentage of protected land results in eight possible categories of hotspot watersheds (Table 3). Ranking according to the level of threat is an intuitive use of the categorization. Although it is likely that the ranks would be sensitive to the person assigning them, it is plausible that the categories with less than the median amount of protected land and at least a 1% increase in the amount of urban land would be identified as the most threatened (categories 4 & 8 in Table 3). There are seven watersheds in the category for the group with drinking water intakes and none in this group for watersheds without drinking water intakes (Fig. 1).

The number of at-risk species and the total number of species are correlated, but not perfectly so, and the imperfect correlation is evident when maps of the two measurements are compared (Figs S2 and S3). Many of the same watersheds are in the 90th percentile for at-risk and total number of species, and this commonality is concentrated in the southeastern United States. The most notable difference between the two occurs in California and the desert southwest. For the number of at-risk species (Fig. S2), there are numerous

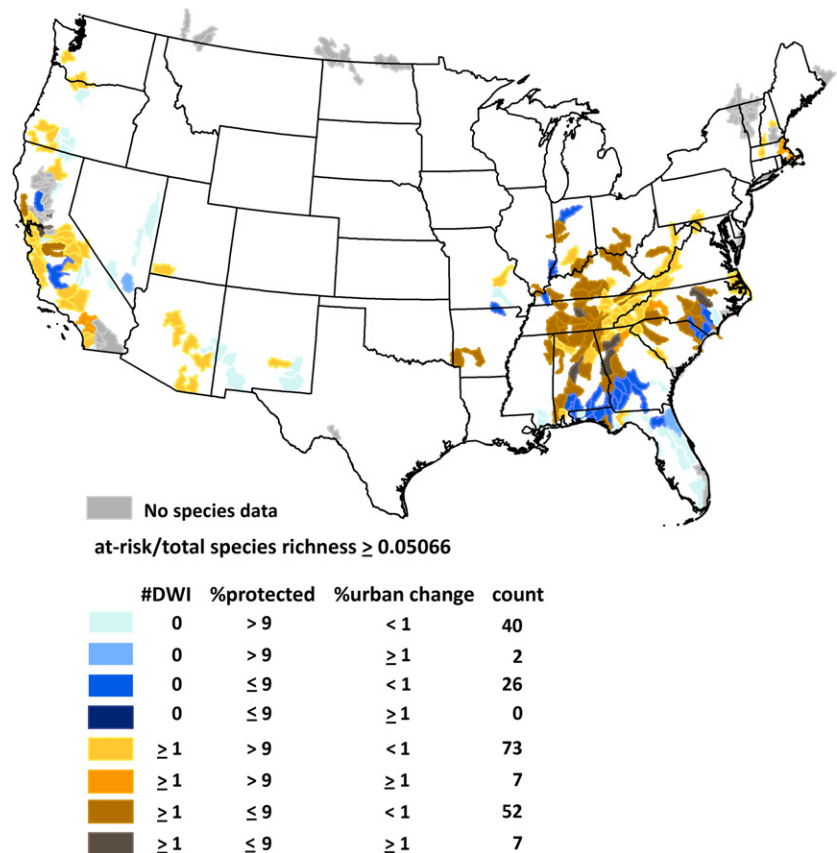


Figure 1 Watersheds in the 90th percentile for the ratio of at-risk species to total species richness categorized by the number of drinking water intakes, percentage of protected land in the watershed and percentage of urban increase in the watershed. The legend lists the thresholds used (see Methods) for number of drinking water intakes, percentage of protected land and percentage of urban increase. Count is the number of watersheds in each category. The 90th percentile for the ratio is 0.05066.

hotspot watersheds in California and the desert southwest, whereas hotspot watersheds for the total number of species are strongly concentrated in south-eastern United States (Fig. S3). The spatial pattern for proportion of at-risk species (Fig. 1) more closely matched the spatial pattern of the number of at-risk species (Fig. S2) than the spatial pattern of total species richness (Fig. S3).

Restricting the analyses to only those species groups for which there were data for all watersheds (plants, birds, mammals, reptiles and amphibians) did not result in dramatic differences in the spatial pattern of hotspot watersheds (Fig. 2). The spatial patterns of species hotspot watersheds using all data (Fig. 2a) and only the species groups for which there were data for all watersheds (Fig. 2b) were similar. Approximately 61% of species hotspot watersheds identified using all data (Fig. 2a) were also species hotspot watersheds using only the species groups for which there were data for all watersheds (Fig. 2b). None of watersheds that were species hotspots in Fig. 2a but not species hotspots in Fig. 2b were watersheds with missing data (Fig. S1), suggesting that changes in the relative proportions of at-risk species for the

species groups without missing data (plants, birds, mammals, reptiles and amphibians) were responsible for the differences between the two maps.

DISCUSSION

We identified numerous geographic locations where biodiversity and drinking water protection are likely to have a shared interest in land conservation. More than 50% of the hotspot watersheds also have drinking water intakes. Adding percentage of protected land and urbanization provided additional thematic resolution that could be used to inform land conservation planning. There were greater numbers of hotspot watersheds with drinking water intakes and less than the median percentage of protected land than hotspot watersheds without drinking water intakes and less than the median percentage of protected land. Further, nearly all of the hotspot watersheds with less than the median percentage of protected land and a high rate of urbanization also had drinking water intakes. The concentration of hotspot watersheds in the south-eastern United States is consistent

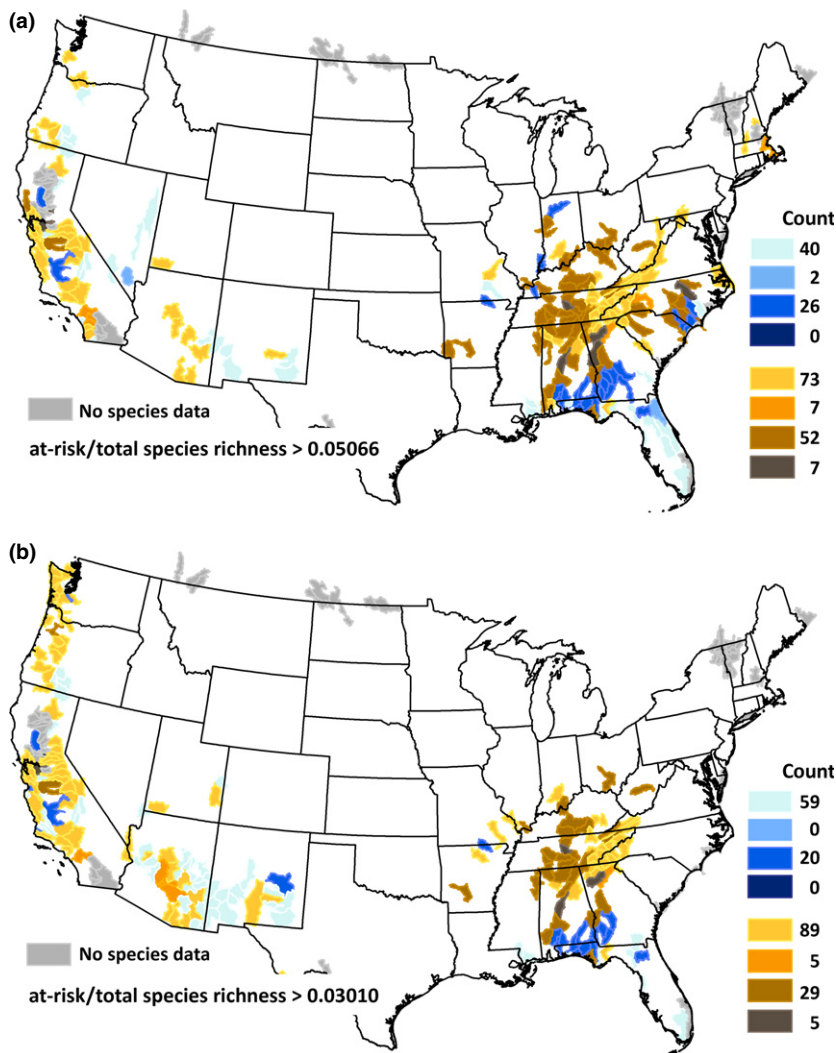


Figure 2 Comparison of hotspot watersheds using the ratio of the number of at-risk species divided by total species richness for all species groups (a) and using only the species groups for which there were data for all watersheds (b). The hotspot watersheds are grouped by the categories listed in Table 3 of the main text, where the top (light blue) colour is category 1 and the bottom (grey) colour is category 8. Figure 2a is the same as Fig. 1. The 90th percentiles were 0.05066 for panel a and 0.03010 for panel b.

with the results reported by Flather *et al.* (1998, 2008) and Robles *et al.* (2008).

The integration of biodiversity and drinking water protection goals that we demonstrate through geographic analysis also applies to the underwriting mechanisms used by each perspective for land conservation. The motivation for the conceptual extension is that the cost of land is high, and the financial resources for land conservation are scarce. For example, the Land and Water Conservation Fund is an important source of underwriting for acquiring lands to protect biodiversity, but only a fraction of the authorized appropriation is made available in any given year (Bean, 2000). Similarly, a US Government Accounting Office review of the US Fish and Wildlife Service National Wildlife Refuge System (NWRS) found that funding was insufficient to manage both the current network of NWRS lands and also to add land to the existing base (see Davison *et al.*, 2006:97). The primary US federal mechanism for funding land conservation for drinking water protection faces similar fiscal constraints. The 1996 amendments to SDWA established the Drinking Water State Revolving Fund (US Environmental Protection Agency, 1997), but only a small portion of these funds are available for land acquisition because the funding mechanism must also support maintenance and development of an extensive drinking water infrastructure (e.g. pipes, water treatment plants; US Environmental Protection Agency,

2009). Integrating the funding mechanisms of each perspective has the potential to leverage scarce resources to conserve land that protects both biodiversity and drinking water.

An example of where integration of these funding mechanisms might occur in geographic space is shown in Fig. 3. The watershed shown in Fig. 3 is one of the seven watersheds in category 8 (Table 3; Fig. 1). The watershed is species rich (proportion of at-risk species = 0.056306), includes drinking water intakes, has little conserved land (~1.5%) and it has a high rate of urbanization (~1.5%). The box in the northwest portion of the watershed bounds Falls Lake, the primary drinking water source for the city of Raleigh, North Carolina. A high proportion of the land surrounding Falls Lake is not identified as conserved in the protected areas database. The Neuse River drains from Falls Lake and serves as a drinking water source for other cities in North Carolina that are downstream from Raleigh. The box in the southeast portion of the watershed bounds a large wetland complex, through which the Neuse River flows, that also has little land that is identified as conserved. Additional conservation of land in the vicinity of Falls Lake and the large wetland complex is likely to benefit both drinking water and biodiversity protection goals.

A watershed east of California's San Francisco Bay (Fig. 4) provides another example of where funding integration might occur. The watershed is also one of the seven

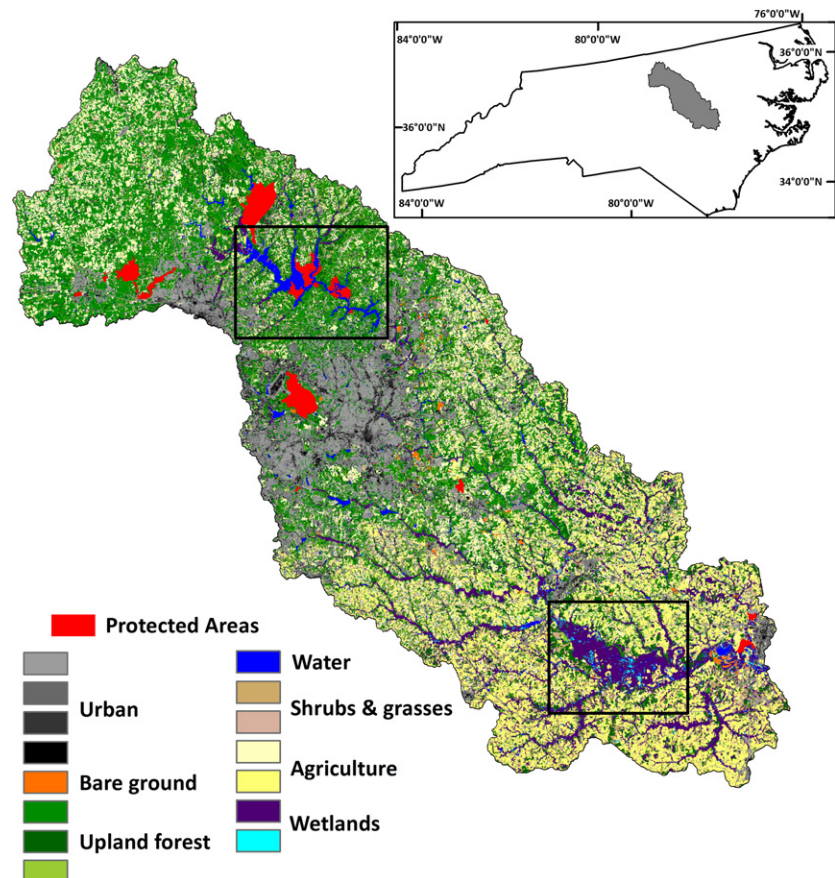


Figure 3 Raleigh, NC watershed. Raleigh is located at the centre of the watershed. The black rectangle northwest of Raleigh bounds Falls Lake, a primary source of drinking water for Raleigh. The Neuse River drains from Falls Lake and serves as a source of drinking water for many downstream cities. The black rectangle at the southern end of the watershed bounds a large wetland complex, that is, part of the Neuse River. The western edge of the city of Goldsboro, NC, is just to the east of the large wetland complex at the watershed boundary. The land cover is from the NLCD 2006 database (<http://www.mrlc.gov/>).

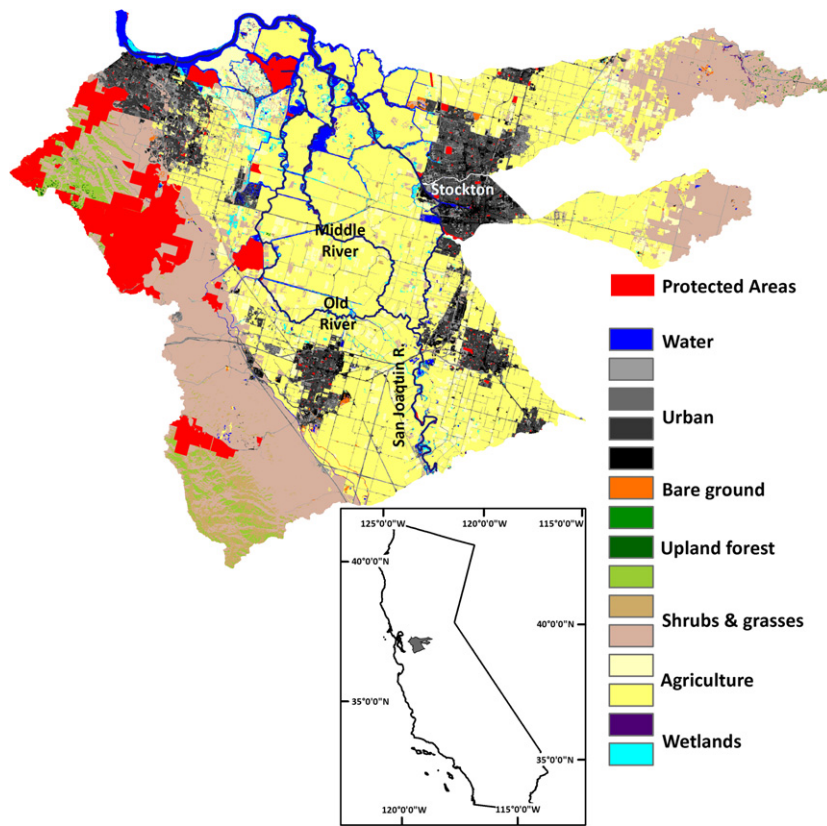


Figure 4 Stockton, CA watershed. The land cover is from the NLCD 2006 database, and the major streams are from the 1:100,000-scale National Hydrography Dataset (NHD; <http://nhd.usgs.gov>).

watersheds in category 8 (Table 3; Fig. 1). It includes drinking water intakes, has a high rate of urbanization (~ 1.3%), and, based on our indicator, proportion of at-risk species, the watershed is in the top 2% for aquatic-dependent biologic diversity in the contiguous United States. Moreover, the watershed flows directly into the San Francisco Bay–Delta estuary – an area that has remained biologically diverse despite substantial anthropogenic modification, supplying habitat to several federally listed threatened or endangered species [e.g. Chinook salmon (*Oncorhynchus tshawytscha*), delta smelt (*Hypomesus transpacificus*), green sturgeon (*Acipenser medirostris*)] and drinking water to ~ 25 million residents throughout the state (National Research Council, 2012). The amount of conserved land (~ 7.4%) is near the United States median of ~ 9%, but there is little protected land in the vicinity of the watershed’s three main rivers. There are several small wetlands in the vicinity of the three main rivers, especially along the southern (upstream) reach of the San Joaquin River, where funding mechanisms for habitat conservation and drinking water protection could come together to synergistically promote biodiversity and clean drinking water.

One advantage of integrating drinking water and biodiversity data is that it adds geographic specificity. Geographic analysis of hotspots alone has been criticized because it identifies areas for conservation that are at a geographic scale, that is, much larger than land managers use when making decisions on conservation (Harris *et al.*, 2005). The narrative

analyses for Figs 3 & 4 do not identify specific 100 ha parcels for conservation, but they do scale down large watersheds (> 300,000 ha) to much smaller areas where more geographically focused analyses (e.g. Davis *et al.*, 2006; Stoms *et al.*, 2011) could be implemented. Another potential advantage of integrating drinking water with biodiversity is a higher likelihood of stakeholder buy-in. Stakeholder buy-in in land conservation for biodiversity protection is often absent or dispassionate because societal needs are often in conflict with the restricted use of the land that conservation imposes (e.g. Thompson, 2006; Díaz-Caravantes & Scott, 2010). Addressing biodiversity conservation through drinking water preservation may have a higher likelihood of stakeholder buy-in because all stakeholders have a tangible interest in clean drinking water. A third potential advantage of integrating drinking water with biodiversity is the potential for stakeholders to recognize the importance of managing the landscape outside areas set aside for conservation. Over-reliance on the land conservation approach to biodiversity protection has been criticized for fostering a dichotomous world of cities and farms that are walled off from ‘outdoor natural heritage museums,’ and, in the end, this may not be the most effective approach to biodiversity conservation (Thompson, 2006; Phalan *et al.*, 2011). Within the realm of water use and conservation, there is an emerging vision that the views of water as a commodity versus water as resource that needs conservation and stewardship are compatible rather than competing (Thompson, 2011). The same concept

intuitively applies to land (Thompson, 2006) as well as water, and this view is implicitly recognized in the SDWA 1996 amendments through the requirement that states submit watershed management plans for drinking water watersheds (US Environmental Protection Agency, 1997). These plans must account for the variety of land uses in the watershed and their potential impact on drinking water security. It is probably true that these state-level plans do not formally incorporate management objectives related to biodiversity preservation, but it would likely be straightforward for stakeholders to incorporate this collateral objective into the plans.

The analyses we presented identified areas where land conservation is likely to support drinking water security and biodiversity protection by applying a simple dichotomy (with or without surface drinking water intakes) to biodiversity hotspot analyses. It is plausible and certainly expected (see, Naidoo & Ricketts, 2006) that positive ecological benefits in addition to improved drinking water security and biodiversity protection could be realized if such land conservation were to occur. For example, acquisition of lands to conserve riparian wetlands could also maintain or improve water quality, contribute to flood mitigation or provide increased opportunities for recreational fishing and waterfowl hunting (de Groot *et al.*, 2006). Our focus on drinking water intakes and at-risk species concentrations was intentionally limited to illustrate how conservation planners could identify watersheds that benefit both human well-being and biodiversity conservation.

ACKNOWLEDGEMENTS

The US Environmental Protection Agency through its Office of Research and Development partially funded and collaborated in the research described here. It has been subjected to agency review and approved for publication. Approval does not signify that the contents reflect the views of the Agency. We also wish to thank Jason McNees, Conservation Data Analyst, for his assistance in querying the NatureServe databases and Megan Mehaffey and the anonymous reviewers for their thoughtful comments on earlier versions of the paper.

REFERENCES

- Ball, I.R., Possingham, H.P. & Watts, M.E. (2009) Marxan and relatives: software for spatial conservation and prioritization. *Spatial conservation prioritization: quantitative methods and computational tools* (ed. by A. Moilanen, K.A. Wilson and H.P. Possingham), pp. 185–195. Oxford University Press, Oxford, UK.
- Bean, M.J. (2000) Strategies for biodiversity protection. *Precious heritage: the status of biodiversity in the United States* (ed. by B.A. Stein, L.S. Kutner and J.S. Adams), pp. 255–273. Oxford University Press, Oxford.
- Beier, P., Majka, D.R. & Spencer, W.D. (2008) Forks in the road: choices in procedures for designing wildland linkages. *Conservation Biology*, **22**, 836–851.
- Beissinger, S.R., Nicholson, E. & Possingham, H.P. (2009) Application of population viability analysis to landscape conservation and planning. *Models for planning wildlife conservation in large landscapes* (ed. by J.J. Millsbaugh and F.R. Thompson III), pp. 33–49. Academic Press, Amsterdam, NL.
- Benedict, M.A. & McMahon, E. (2006) *Green infrastructure: linking landscapes and communities*. Island Press, Washington, DC, USA.
- Carroll, C., Noss, R.E., Paquet, P.C. & Schumaker, N.H. (2003) Use of population viability analysis and reserve selection algorithms in regional conservation plans. *Ecological Applications*, **13**, 1773–1789.
- Davis, F.W., Costello, C. & Stoms, D. (2006) Efficient conservation in a utility-maximization framework. *Ecology and Society*, **11**, 33. [online] Available at: URL: <http://www.ecologyandsociety.org/vol11/iss1/art33/> (accessed 4 June 2013).
- Davison, R.P., Falcucci, A., Maiorano, L. & Scott, J.M. (2006) The National Wildlife Refuge System. *The Endangered Species Act at thirty: renewing the conservation promise*. Volume 1. (ed. by D.D. Goble, J.M. Scott and F.W. Davis), pp. 90–100. Island Press, Washington, DC, USA.
- DellaSala, D.A., Staus, N.L., Strittholt, J.R., Hackman, A. & Iacobelli, A. (2001) An updated protected areas database for the United States and Canada. *Natural Areas Journal*, **21**, 124–135.
- Díaz-Caravantes, R.E. & Scott, C.A. (2010) Water management and biodiversity conservation interface in Mexico: a geographical analysis. *Applied Geography*, **30**, 343–354.
- Dudley, N. & Stolton, S. (2003) Running pure: the importance of forest protected areas to drinking water. World Bank/WWF Alliance for Forest Conservation and Sustainable Use. Available at: <http://assets.panda.org/downloads/runningpurereport.pdf>. (accessed 4 June 2013).
- Ernst, C., Gullick, R. & Nixon, R. (2004) Protecting the source: conserving forests to protect water. *Opflow*, **30**, 1, 4–7.
- Flather, C.H., Knowles, M.S. & Kendall, I.A. (1998) Threatened and endangered species geography. *BioScience*, **48**, 365–376.
- Flather, C.H., Knowles, M.S. & McNees, J. (2008) *Geographic patterns of at-risk species*. RMRS-GTR-211. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. Fort Collins, CO, USA.
- Flather, C.H., Wilson, K.R. & Shriner, S.A. (2009) Geographic approaches to biodiversity conservation: implications of scale and error to landscape planning. *Models for planning wildlife conservation in large landscapes* (ed. by J.J. Millsbaugh and F.R. Thompson III), pp. 85–121. Academic Press, Amsterdam, NL.
- Fry, J.A., Xian, G., Jin, S., Dewitz, J.A., Homer, C.G., Yang, L.M., Barnes, C.A., Herold, N.D. & Wickham, J.D. (2011) Completion of the 2006 National Land Cover Database for the conterminous United States. *Photogrammetric Engineering and Remote Sensing*, **77**, 858–864.

- de Groot, R., Stuij, M., Finlayson, M. & Davidson, N. (2006) *Valuing wetlands: guidance for valuing the benefits derived from wetland ecosystem services*. Ramsar Convention, Gland, Switzerland.
- Harris, G.M., Jenkins, C.N. & Pimm, S.L. (2005) Refining biodiversity conservation priorities. *Conservation Biology*, **19**, 1957–1968.
- Loftus, A.J. & Flather, C.H. (2012) *Fish and other aquatic resource trends in the United States: a technical document supporting the Forest Service 2010 RPA assessment*. RMRS-GTR-283. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA.
- McNees, J. (2010) Metadata for the U.S. Forest Service – Rocky Mountain Research Station: 2010 RPA biodiversity datasets. NatureServe Central Databases.
- Mehaffey, M.H., Nash, M.S., Wade, T.G., Ebert, D.W., Jones, K.B. & Rager, A. (2005) Linking land cover and water quality in New York City's water supply watersheds. *Environmental Monitoring and Assessment*, **107**, 29–44.
- Minor, E.S. & Urban, D.L. (2008) A graph-theory framework for evaluating landscape connectivity and conservation planning. *Conservation Biology*, **22**, 297–307.
- Muneepeerakul, R., Bertuzzo, E., Lynch, H.J., Fagan, W.F., Rinaldo, A. & Rodriguez-Iturbe, I. (2008) Neutral meta-community models predict fish diversity patterns in Mississippi-Missouri basin. *Nature*, **453**, 220–222.
- Naidoo, R. & Ricketts, T.H. (2006) Mapping the economic costs and benefits of conservation. *Plos Biology*, **4**, e360. doi:10.1371/journal.pbio.0040360.
- National Research Council (2000) *Watershed management for potable water supply: assessing the New York City strategy*. The National Academy Press, Washington, DC, USA.
- National Research Council (2005) *Valuing ecosystem services: toward better environmental decision-making*. The National Academies Press, Washington, DC, USA.
- National Research Council (2012) *Sustainable water and environmental management in the California Bay-Delta*. The National Academies Press, Washington, DC, USA.
- Phalan, B., Onial, M., Balmford, A. & Green, R.E. (2011) Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science*, **333**, 1289–1291.
- Possingham, H.P., Wilson, K.A., Aldelman, S.J. & Vynne, C.H. (2006) Protected areas: goals, limitations, and design. *Principles of conservation biology* (ed. by M.J. Groom, G.K. Meffe and C.R. Carroll), pp. 507–549. Sinauer Associates, Sunderland, MA, USA.
- Postel, S.L. & Thompson, B.H., Jr (2005) Watershed protection: capturing the benefits of nature's water supply services. *Natural Resources Forum*, **29**, 98–108.
- Robles, M.D., Flather, C.H., Stein, S.M., Nelson, M.D. & Cutko, A. (2008) The geography of private forests that support at-risk species in the conterminous United States. *Frontiers in Ecology and the Environment*, **6**, 301–307.
- Rosenzweig, M.L., Donoghue, J., Li, Y.M. & Yuan, C. (2011) Estimating species density. *Biological diversity: frontiers in measurement and assessment* (ed. by A.E. Magurran and B.J. McGill), pp. 276–288. Oxford University Press, Oxford, UK.
- Scott, J.M., Csuti, B., Jacobi, J.D. & Estes, J.E. (1987) Species richness – a geographic approach to protecting future biological diversity. *BioScience*, **37**, 782–788.
- Scott, J.M., Davis, F., Csuti, B., Noss, R., Butterfield, B., Groves, C., Anderson, H., Caicco, S., D'Erchia, F., Edwards, Jr, T.C., Ulliman, J. & Wright, R.G. (1993) Gap analysis: a geographic approach to protection of biological diversity. *Journal of Wildlife Management*, **123**, 3–41.
- Shaffer, M.L., Scott, J.M. & Casey, F. (2002) Noah's options: initial cost estimates of a national system of habitat conservation areas in the United States. *BioScience*, **52**, 439–443.
- Shriner, S.A., Wilson, K.R. & Flather, C.H. (2006) Reserve networks based on richness hotspots and representation vary with scale. *Ecological Applications*, **16**, 1660–1673.
- Stoms, D.M., Kreidler, J. & Davis, F.W. (2011) The power of information for targeting cost-effective conservation investments in multifunctional farmlands. *Environmental Modelling & Software*, **26**, 8–17.
- Thompson, B.H., Jr (2006) Managing the working landscape. *The Endangered Species Act at thirty: renewing the conservation promise*. Volume 1. (ed. by D.D. Goble, J.M. Scott and F.W. Davis), pp. 101–126. Island Press, Washington, DC, USA.
- Thompson, B.H., Jr (2011) Water as a public commodity. *Marquette Law Review*, **95**, 17–52.
- U.S. Environmental Protection Agency (1997) *State source water assessment and protection program: final guidance*. EPA 816-R-97-009. U.S. EPA Office of Water, Washington, DC, USA.
- U.S. Environmental Protection Agency (2009) *Drinking water state revolving fund 2009 annual report*. EPA 816-R-10-021. U.S. EPA Office of Water, Washington, DC, USA.
- Watts, M.E., Ball, I.R., Stewart, R.S., Klein, C.J., Wilson, K., Steinback, C., Lourival, R., Kircher, L. & Possingham, H.P. (2009) Marxan with Zones: software for optimal conservation based land- and sea-use zoning. *Environmental Modelling & Software*, **24**, 1513–1521.
- Wickham, J.D., Riitters, K.H., Wade, T.G. & Vogt, P. (2010) A national assessment of green infrastructure and change for the conterminous United States using morphological image processing. *Landscape and Urban Planning*, **94**, 186–195.
- Wickham, J.D., Wade, T.G. & Riitters, K.H. (2011) An environmental assessment of United States drinking water watersheds. *Landscape Ecology*, **26**, 605–616.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Figure S1 Watersheds without data (light blue) for non-vascular plants, fungi and lichens, invertebrates and freshwater fish. Watersheds with no data are grey.

Figure S2 Watersheds in the 90th percentile for number of at-risk species. The 90th percentile value for at-risk species was 20.

Figure S3 Watersheds in the 90th percentile for total species richness. The 90th percentile value for total species richness was 437.

BIOSKETCHES

Jim Wickham is a senior research biologist with the National Exposure Research Laboratory in Research Triangle Park, NC. His research interests include land cover mapping, assessment of impaired waters and landscape pattern analysis.

Curt Flather is a research ecologist with the Rocky Mountain Research Station in Fort Collins, CO, where his research is focused on understanding biodiversity response to changing landscape patterns driven by climate change, land use, natural disturbance and land management activities.

Author contributions: J.W. designed the study, conducted the analysis and wrote and edited the paper; C.F. conducted the analysis and wrote and edited the paper.

Editor: Andrew Knight