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[Management Brief]

Power of Revisit Monitoring Designs to Detect Forestwide Declines in Trout Populations

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Abstract.—Managers are often required to monitor networks of sites to make inferences about trends in fish populations over large geographic areas. We evaluated the statistical power of four revisit monitoring designs (always revisit design; two types of augmented serially alternating design; and serially alternating design) to detect declines in trout population biomass over a 30-year time horizon. The Medicine Bow National Forest, Wyoming, was used as a case study, and the designs were based on sampling 20 or 30 sites every other year but varied in the total number of sites (20–40 or 30–60) and the timing and frequency with which sites were monitored. The always revisit design, in which the same 20 or 30 sites were sampled during every monitoring period, often had slightly higher power than the other designs. However, any differences in power quickly diminished due to the rapid increase in power over time for all four designs. The similar levels of statistical power to detect population declines among the designs we evaluated suggest that managers have flexibility to choose a revisit design with increased spatial coverage and implementation complexity without sacrificing trend detection capability.

Managers monitor aquatic habitats and fish populations to assess population status and trends in response to management actions, human disturbances, or exogenous factors such as climate change (Gibbs et al. 1998). Management plans developed by land management agencies and fish and wildlife agencies often include monitoring protocols intended to yield data that will guide management decisions and withstand legal challenges (Murphy and Noon 1991; Urquhart et al. 1998).

Although managers may monitor single sites that are relevant to specific fisheries, oftentimes a network of sites selected probabilistically is required to make inferences across stream systems or large geographic areas (Gibbs et al. 1998; Larsen et al. 2001; Brown et al. 2005). For example, the Oregon Department of Fish and Wildlife monitors a network of sites to make statewide inferences regarding the status and trends of salmon populations and habitats (Firman and Jacobs 2001). The U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program uses a rotating panel design to monitor a network of stream sites to estimate the status and trend of aquatic resources across the United States (Stevens 1994; Herlihy et al. 2000).

The statistical power ($1 - \beta$, where β = type II statistical error) to detect trends in a network of sites is dependent on several factors. Similar to detecting trends at a single site, power for a network of sites is influenced by the variance of the trend estimate, the sample size, the magnitude of trend (effect size), the direction of trend (positive or negative), and the type I statistical error (Gerrodette 1987; Gerow 2007). When monitoring a network of sites, sample size includes both the number of sites monitored and the number of revisits to those sites across time. Power is also influenced by the monitoring design (i.e., the timing and frequency with which sites are sampled over time; Urquhart and Kincaid 1999; Larsen et al. 2004).

There are many survey designs available to managers when monitoring a network of sites (Urquhart and Kincaid 1999; McDonald 2003). Some designs specify revisiting one panel of the same sites during every year in which monitoring occurs. Other designs specify revisiting some panels of sites multiple times while sampling others only once. Still other designs specify sampling all sites only once, with different panels of sites being sampled during each monitoring year. Within these general designs, the timing and frequency of monitoring for the panels of sites can vary across the monitoring time frame. Designs where one panel of sites in the network is sampled in each monitoring year have the highest statistical power to detect change over time at those sites, but generalization to other sites can be problematic when a small total number of sites is monitored (Larsen et al. 2004). By contrast, designs

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that integrate more sites into the network but alternate among sites across sampling periods have lower power to detect trends but provide better spatial coverage (Urquhart and Kincaid 1999; Quist et al. 2006; Dauwalter et al. 2009).

In the United States, national forests are required to monitor management indicator species to determine the forestwide impacts of management on aquatic ecosystems and to aid in forest planning (Hayward et al. 2001). Our objective was to evaluate the statistical power of four different revisit monitoring designs to detect forestwide declines in trout population biomass over a 30-year time horizon. All four designs involved revisiting the same sites over time, but the total number of sites and the timing and frequency of sampling were varied to evaluate the tradeoff between trend detection capability (statistical power) and spatial coverage (total number of unique sites). We evaluate our objective by using the Medicine Bow National Forest in southeastern Wyoming as a case study.

Methods

The Medicine Bow National Forest encompasses 5,600 km^2 across four units: Sierra Madre, Snowy Range, Laramie Peak, and Pole Mountain. Land cover consists primarily of lodgepole pine *Pinus contorta*, Engelmann spruce *Picea engelmannii*, and subalpine fir *Abies lasiocarpa* but also includes sage steppe at lower elevations around the forest periphery. Streamflow patterns are typical of snowmelt streams, with peak flows in June. The Colorado River cutthroat trout *Oncorhynchus clarkii pleuriticus* is the only trout native to this forest but is only native to streams draining west of the Continental Divide on the westernmost Sierra Madre unit. Streams are primarily occupied by naturalized populations of brook trout *Salvelinus fontinalis* and brown trout *Salmo trutta*, but rainbow trout *O. mykiss* also occur in some streams.

Monitoring protocol.—Monitoring objectives were set during planning meetings between biologists from the Medicine Bow National Forest and researchers at the University of Wyoming. The sampling frame for monitoring in the forest consisted of 445 eighth-level hydrologic unit code watersheds (Verdin and Verdin 1999) that (1) had less than 50% alpine–subalpine land cover; (2) had one or more streams present at the 1:100,000 scale on forest land (nonprivate land); and (3) had less than 50% private land. These criteria eliminated stream sites at elevations too high to support fish populations and also eliminated sites on private land, where the U.S. Forest Service (USFS) would have little opportunity to manage resources. The Medicine Bow National Forest selected all trout collectively (brook trout, rainbow trout, and brown trout) as its aquatic management indicator taxon. The objectives identified for monitoring were to detect a 2.5% annual decline in trout biomass (kg/ha) over a 30-year time horizon with 80% statistical power ($1 - \beta = 0.80$) at a significance level (α) equal to 0.20. These levels of significance and power were chosen to set equivalent our type I (detecting false trends) and type II (failure to detect trends that are real) statistical error rates. We did not blindly adopt the more common but arbitrary α values (i.e., 0.05 or 0.10) since we considered the management consequences of failing to detect real changes in trout populations to be more important than detecting a false trend (Gibbs et al. 1998; Maxell 1999). Sampling units were defined as the individual eighth-level hydrologic unit code watersheds, and trends in trout biomass were to be measured in a 150-m stream reach within each watershed. Logistical and financial considerations constrained monitoring to occur at only 30 sites during each monitoring year, with field sampling to occur every other year. Since there is the potential for reduced financial support for monitoring in the future, we alternatively planned for monitoring to occur at 20 sites during each monitoring year, with field sampling to occur every other year.

Monitoring designs.—We evaluated the statistical power of four revisit monitoring designs to detect declines in trout biomass over time: an always revisit design, two augmented serially alternating designs, and a serially alternating design (cf. Urquhart and Kincaid 1999). For the scenario with 30 sites/monitoring period, the always revisit design consisted of one panel of 30 sites sampled in each monitoring period (every 2 years; Table 1). The first augmented serially alternating design consisted of 40 unique sites and involved sampling one panel of 20 sites in each monitoring period (every 2 years) and alternating between two panels of 10 sites every other monitoring period so that each of these panels was resampled every 4 years. The second augmented serially alternating design consisted of 50 unique sites and involved sampling one panel of 10 sites in each monitoring period and alternating between two panels of 20 sites every other monitoring period so that each of these panels was resampled every 4 years. The serially alternating design consisted of 60 unique sites and involved alternating between two panels of 30 sites every other monitoring period (each panel was sampled every 4 years). For the scenario of 20 sites/monitoring period, the monitoring designs were the same but the total number of sites per design varied from 20 to 40, and the number of sites in each panel ranged from 5 to 20 (Table 1).

TABLE 1.—Four revisit monitoring designs evaluated for their ability to detect declines in trout population biomass in the Medicine Bow National Forest, Wyoming. Sampling was limited to 20 or 30 sites/year, where monitoring would occur every other year (always revisit design = always revisit the same sites during each monitoring period; augmented serially alternating designs 1 and 2 = always revisit one panel of sites each period and alternate two panels of sites every other period; serially alternating design = alternate two panels of sites in every period). An "x" indicates the years when sites in each panel are sampled within each monitoring design.

		Sites/year		Elapsed time since initial monitoring period (years)								
Design	Panel	30	20	0 (initial)	2	4	6	8	10	. . .	28	30
Always revisit	1	30	20	x	x	x	x	x	x	. . .	x	x
Augmented serially alternating 1	1	20	15	x	x	x	x	x	x	. . .	x	x
	2	10	5	x		x		x		. . .	x	
	3	10	5		x		x		x	. . .		x
Augmented serially alternating 2	1	10	10	x	x	x	x	x	x	. . .	x	x
	2	20	10	x		x		x		. . .	x	
	3	20	10		x		x		x	. . .		x
Serially alternating	1	30	20	x		x		x		. . .	x	
	2	30	20		x		x		x	. . .		x

Simulated statistical power.—We estimated the statistical power of each monitoring design to detect declines in trout populations by using Monte Carlo simulations (Figure 1; Manly 1997; Gibbs et al. 1998). First, we simulated the initial biomass (at time $t = 0$) for all sites in each network using a $\log_e$ normal distribution ($\log_e$ normal mean biomass = 79.3 kg/ha; SD = 2.9 kg/ha) that was specified using data collected from the Medicine Bow National Forest. We then projected a 2.5% annual decline over 30 years onto each initial biomass estimate per site. Following Gerrodette (1987), we used an exponential growth model in which biomass at time t is proportional to the biomass during the previous time period:

$$M_{st} = M_{s0}(1 + r)^t,$$

where M_{st} is biomass at site s and time t, M_{s0} is the initial biomass at site s, and r is the rate of change ($r = -0.025$) for which 97.5% of the biomass at time t will occur at time ($t + 1$). The log-transformed model is

$$\log_e M_{st} = \log_e M_{s0} + \log_e(1 + r)t$$

and can be estimated using linear least-squares regression:

$$\log_e M_{st} = b_0 + b_1 + \varepsilon,$$

where b_0 is the intercept and is equal to the $\log_e$ of initial biomass ($\log_e M_{s0}$); b_1 is the slope parameter, the antilog of which represents percent annual change in biomass (100% × exp[b_1]); and ε is normally distributed error with a mean of 0 (Gerrodette 1987; Thompson et al. 1998). We added random variation (ε) onto each annual biomass estimate per site per year (t = year 0, 6, 10, 20, and 30). The amount of random error was determined using a coefficient of variation (CV = 100 × SD/mean) in biomass that was unique to each site. The CV for each site was selected from a distribution of CVs in biomass ($\log_e$ normal mean CV = 0.54; SD = 0.021) based on a literature review of temporal variation in biomass of trout populations (Dauwalter et al. 2009). Based on these inputs, the collection of biomasses was generated for each site and year, and this simulation process was repeated 1,000 times.

From the simulated data set, we selected the number of sites and years called for by each monitoring design and evaluated the statistical power of each design to detect the underlying declines in biomass given the spatial and temporal variability in the data set. Following Gibbs et al. (1998), we evaluated statistical power for each design by estimating trend individually for each site using the slope parameter from a linear regression of $\log_e$(biomass) versus year (100% × exp[$\hat{b}_1$] = estimated % annual decline). We then computed the mean slope (trend) across sites and used a one-tailed t-test to determine whether the mean slope across sites was significantly less than zero (H_0: $\hat{\bar{b}}_1 = 0$) using the specified α. The proportion of 1,000 simulations in which a significant trend was detected constituted our estimate of power for each sampling design (sensu Gibbs et al. 1998). Although the monitoring objectives called only for detecting a 2.5% decline in biomass at an α of 0.20, we generalized our results by also evaluating power to detect changes in 1.0, 2.5, and 5.0% annual declines at α values equal to 0.05, 0.10, and 0.20 for years 6, 10, 20, and 30.

Results

All four monitoring designs showed similar statistical power to detect declines in trout population

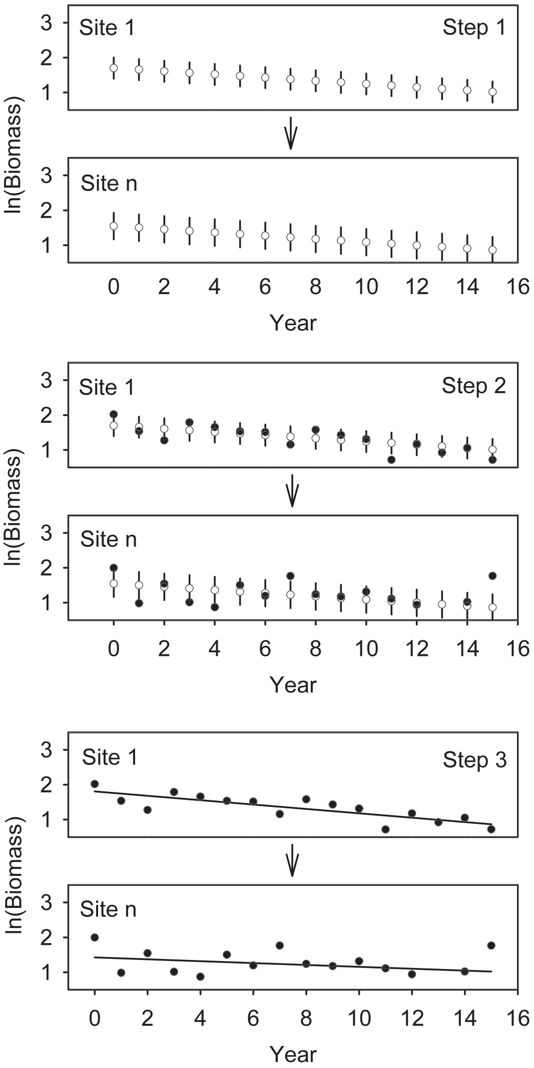

FIGURE 1.—Steps for simulating statistical power of a monitoring design to detect trends in trout biomass at a network of multiple sites. In step 1, a declining trend was projected onto an initial biomass estimate ($\log_e$ transformed) for a site over the projected monitoring time frame. For each year, open circles represent the mean biomass and lines represent 1 SD. In step 2, random variation was added to the biomass estimate for each year (black shaded circles) using a random deviate from a normal distribution, with a mean equal to the projected biomass and an SD approximated for the initial biomass estimate. In step 3, the slope of the least-squares regression line was estimated for each site. We then determined whether the arithmetic mean slope among sites was different from zero based on the specified significance level (α). We repeated each step 1,000 times, and the proportion of times in which the slope differed from zero was used to represent statistical power and to assess how often the known trends were detected amid simulated variation.

biomass over time (Figures 2, 3). The always revisit design often had slightly higher power to detect large population declines during the initial years of monitoring (Figure 2C, F, I; Figure 3C, F, I), but this advantage quickly diminished due to the rapid increase in power over time for all four designs (Figures 2, 3). As expected, there was more power to detect larger annual declines in biomass, and power increased as the number of years of monitoring increased. Consider, for example, the ability to detect a decline at an α of 0.20. When 30 sites were monitored in each year of monitoring, the length of time required to detect a known decline with high power ($\geq$0.80) was 25 years for a 1% annual decline, 14 years for a 2.5% annual decline, and only 8 years for a 5% annual decline (Figure 2G–I). When 20 sites were monitored, the length of time required to detect a known decline with high power ($\geq$0.80) was 29 years for a 1% annual decline, 16 years for a 2.5% annual decline, and 9 years for a 5% annual decline (Figure 3G–I).

Discussion

We showed that four monitoring designs that specify revisiting of the same sites over time have similar statistical power to detect declines in trout population biomass. The similar power among designs was probably due to the tradeoff between the precision of the trend estimate at each site (reduced variance) versus a greater number of trend estimates across sites (increased sample size). The always revisit design specified revisiting all sites during each monitoring year, and this would result in the most precise estimates of trend for each site. However, since the statistical test was based on the mean trend across sites, the gain in power from more precise trend estimates at each site was largely lost because of the reduced number of total sites (sample size) on which the mean trend estimate was based. Others have also shown always revisit designs to have only marginally higher power when compared with augmented serially alternating designs (Urquhart and Kincaid 1999).

Although there were mostly negligible differences in power among the designs we evaluated, the design ultimately selected by the Medicine Bow National Forest for implementation was the always revisit design. The always revisit design was chosen because it would be the simplest to implement. It also contained the fewest sites and therefore would require the least amount of initial site reconnaissance and setup time. Forest biologists also conveyed that implementing the least complex design would also increase the likelihood that monitoring would continue over the duration of the monitoring time frame even in the event of staff turnover.

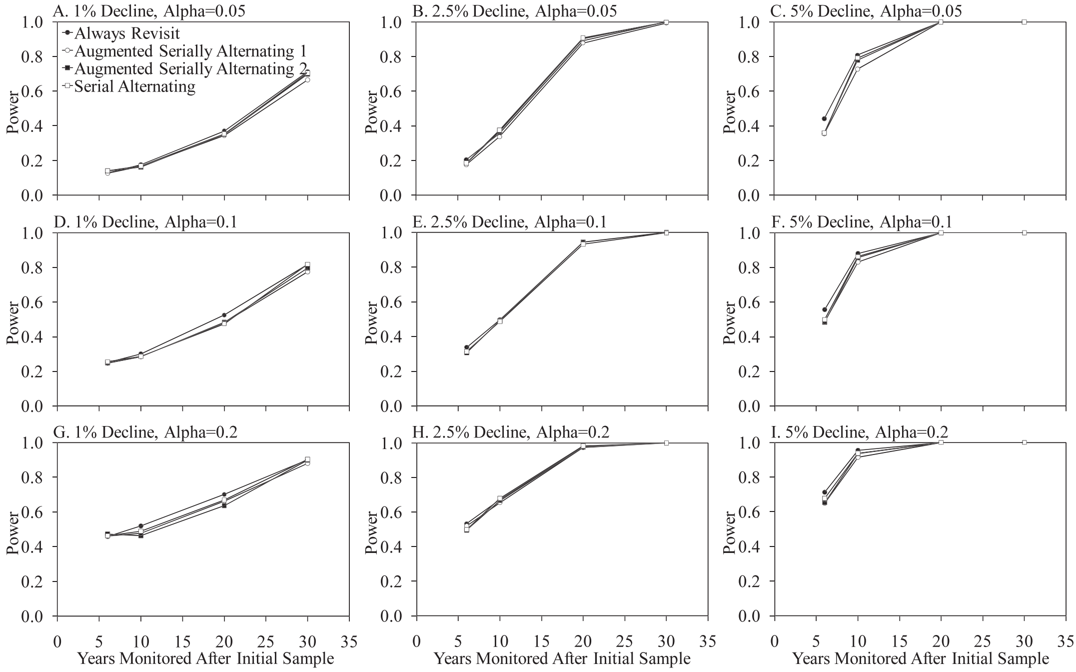

FIGURE 2.—Statistical power to detect 1.0, 2.5, and 5.0% annual declines in trout population biomass at significance levels (α) of 0.05, 0.10, and 0.20 under four monitoring designs evaluated for the Medicine Bow National Forest, Wyoming. Sampling was limited to 30 sites/year, where monitoring would occur every other year (always revisit design = always revisit the same 30 sites in each monitoring period; augmented serially alternating design 1 = always revisit 20 sites in each period and alternate between two panels of 10 sites; augmented serially alternating design 2 = always revisit 10 sites and alternate between two panels of 20 sites; serially alternating design = alternate between two panels of 30 sites in every period).

Others implementing monitoring protocols may choose to employ more sophisticated monitoring designs that incorporate more sites and better spatial coverage without sacrificing trend detection capability. The serially alternating design we evaluated for the Medicine Bow National Forest contained the most unique sites and would have had better spatial coverage of trout populations while still maintaining the power to detect population trends at a level similar to the power offered by the other, less-complex, designs. Larsen et al. (2004) showed that revisit designs were relatively insensitive to the total number of sites in a monitoring network, but they suggested that 30–50 sites be used to ensure adequate spatial representation of regional-scale trends. The Oregon Department of Fish and Wildlife uses a complex panel design to monitor salmon populations and habitats, where some panels of sites are revisited and others are sampled only once (Firman and Jacobs 2001). The complexity of that design allows them to meet their dual objectives of monitoring the status and trends of populations and habitats.

Our power calculations could be conservative. We assumed that biomass across sites was uncorrelated in time. Populations can fluctuate synchronously over time due to large-scale factors (e.g., climate), which can decrease the power to detect population trends (Larsen et al. 2004). However, in exploratory analyses of other published trout population data (Platts and Nelson 1988), we found synchronous variation to represent less than 1% of the total variation observed across sites (D. C. Dauwalter, unpublished data); therefore, our failure to incorporate correlated variation across sites likely had only a small effect on our power estimates. We also assumed a constant 2.5% decline in biomass across sites. Variation in trends across sites will decrease power, but we do not know how spatially variable forest management will be in the future and how population trends will vary in response. Lastly, our estimate of temporal variation was less than that reported by others. Ham and Pearsons (2000) reported CVs from 26% to 94% for salmonid abundance in the Yakima River system, Washington, and suggested that the level of population variation they observed prohibited sufficient feedback to allow for corrective management action. Since we lacked CV estimates

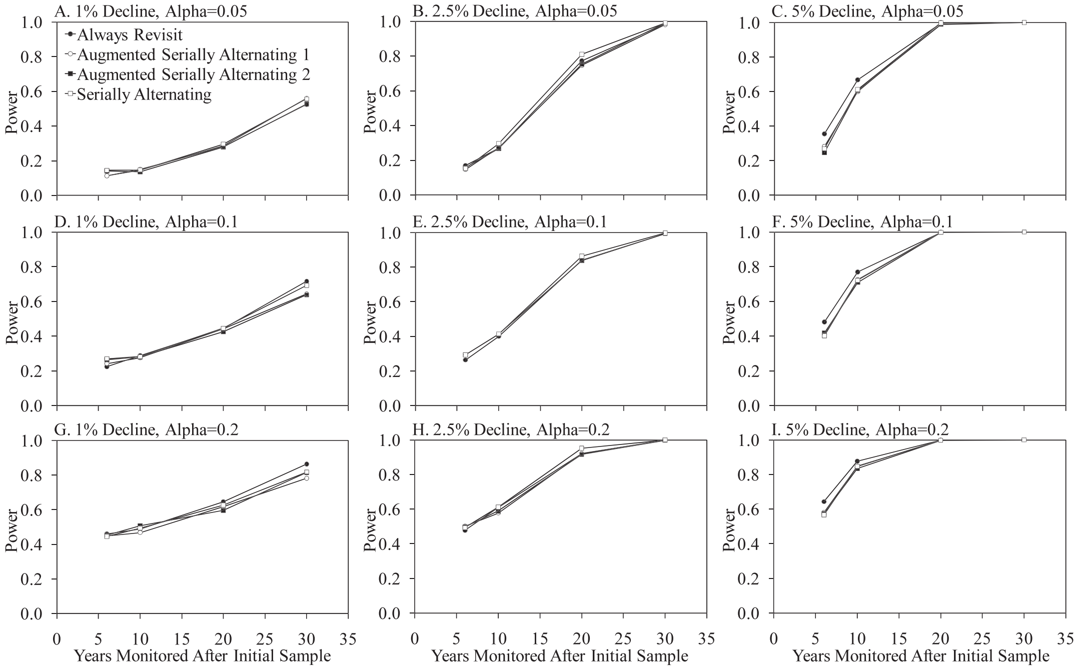

FIGURE 3.—Statistical power to detect 1.0, 2.5, and 5.0% annual declines in trout population biomass at significance levels (α) of 0.05, 0.10, and 0.20 under four monitoring designs evaluated for the Medicine Bow National Forest, Wyoming. Sampling was limited to 20 sites/year, where monitoring would occur every other year (always revisit design = always revisit the same 20 sites in each monitoring period; augmented serially alternating design 1 = always revisit 15 sites in each period and alternate between two panels of five sites; augmented serially alternating design 2 = always revisit 10 sites and alternate between two panels of 10 sites; serially alternating design = alternate between two panels of 20 sites in every period).

from the Medicine Bow National Forest, we used literature-derived CV estimates (including those from Ham and Pearsons 2000) that we thought would sufficiently inform our simulations. As monitoring proceeds, CVs in biomass can be estimated to determine whether our simulations are likely be optimistic, especially since variance estimates used in prospective power analyses can be off by as much as 50% (Carey and Keough 2002). However, the small differences in power among the designs we evaluated should persist even if our power estimates are conservative.

A power analysis is a necessary step during monitoring protocol development to ensure that monitoring can detect the level of population changes stated by the monitoring objectives (Gryska et al. 1997; Beck et al. 2010). We showed that several different revisit designs have similar abilities to detect declines in trout populations of the Medicine Bow National Forest and that any slight differences in power among the revisit designs we evaluated were quickly recouped with a few more years of monitoring. Although we did not explore all possible revisit designs or every combination of variables known to influence power, we evaluated designs that are commonly used and feasible to implement by fisheries biologists (McDonald 2003). Consequently, managers that are developing monitoring protocols based on revisit monitoring designs similar to the ones we evaluated should have flexibility within a revisit design framework to meet their fiscal and logistical needs without sacrificing their ability to detect population changes. However, managers that are considering monitoring designs that vary substantially from the ones we evaluated should complete a prospective power analysis to determine whether power differs enough among designs as to not be quickly recouped with time.

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References

Beck, J. L., D. C. Dauwalter, K. G. Gerow, and G. D. Hayward. 2010. Design to monitor trend in abundance and presence of American beaver (*Castor canadensis*) at the national forest scale. Environmental Monitoring and Assessment 164:463–479.

Brown, B. S., N. E. Detenbeck, and R. Eskin. 2005. How probability survey data can help integrate 305(B) and 303(D) monitoring and assessment of state waters. Environmental Monitoring and Assessment 103:41–57.

Carey, J. M., and M. J. Keough. 2002. The variability of estimates of variance, and its effect on power analysis in monitoring design. Environmental Monitoring and Assessment 74:225–241.

Dauwalter, D. C., F. J. Rahel, and K. G. Gerow. 2009. Temporal variation in trout populations: implications for monitoring and trend detection. Transactions of the American Fisheries Society 138:38–51.

Firman, J. C., and S. E. Jacobs. 2001. A survey design for integrated monitoring of salmonids. Pages 242–252 *in* T. Nishida, P. J. Kailola, and C. E. Hollingworth, editors. Proceedings of the first international symposium on geographic information systems (GIS) in fishery science. Fishery GIS Research Group, Saitama, Japan.

Gerow, K. G. 2007. Power and sample size estimation techniques for fisheries management: assessment and a new computational tool. North American Journal of Fisheries Management 27:397–404.

Gerrodette, T. 1987. A power analysis for detecting trends. Ecology 68:1364–1372.

Gibbs, J. P., S. Droege, and P. Eagle. 1998. Monitoring populations of plants and animals. BioScience 48:935–940.

Gryska, A. D., W. A. Hubert, and K. G. Gerow. 1997. Use of power analysis in developing monitoring protocols for the endangered Kendall Warm Springs dace. North American Journal of Fisheries Management 17:1005–1009.

Ham, K. D., and T. N. Pearsons. 2000. Can reduced salmonid population abundance be detected in time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences 57:17–24.

Hayward, G. D., N. M. Warren, B. Parrish, M. Williams, C. Liggett, and V. Starostka. 2001. Region 2 management indicator species selection process and criteria. USDA Forest Service, Lakewood, Colorado.

Herlihy, A. T., D. P. Larsen, S. G. Paulsen, N. S. Urquhart, and B. J. Rosenbaum. 2000. Designing a spatially balanced, randomized site selection process for regional stream surveys: the EMAP Mid-Atlantic pilot study. Environmental Monitoring and Assessment 63:95–113.

Larsen, D. P., P. R. Kaufmann, T. M. Kincaid, and N. S. Urquhart. 2004. Detecting persistent change in the habitat of salmon-bearing streams in the Pacific Northwest. Canadian Journal of Fisheries and Aquatic Sciences 61:283–291.

Larsen, D. P., T. M. Kincaid, S. E. Jacobs, and N. S. Urquhart. 2001. Designs for evaluating local and regional scale trends. BioScience 61:1069–1078.

Manly, B. F. J. 1997. Randomization, bootstrap and Monte Carlo methods in biology, 2nd edition. Chapman and Hall, London.

Maxell, B. A. 1999. A power analysis on the monitoring of bull trout stocks using redd counts. North American Journal of Fisheries Management 19:860–866.

McDonald, T. L. 2003. Review of environmental monitoring methods: survey designs. Environmental Monitoring and Assessment 85:277–292.

Murphy, D. D., and B. D. Noon. 1991. Coping with uncertainty in wildlife biology. Journal of Wildlife Management 55:773–782.

Platts, W. S., and R. L. Nelson. 1988. Fluctuations in trout populations and their implications for land-use evaluation. North American Journal of Fisheries Management 8:333–345.

Quist, M. C., K. G. Gerow, M. R. Bower, and W. A. Hubert. 2006. Random versus fixed-site sampling when monitoring relative abundance of fishes in headwater streams of the upper Colorado River basin. North American Journal of Fisheries Management 26:1011–1019.

Stevens, D. L. Jr. 1994. Implementation of a national monitoring program. Journal of Applied Management 42:1–29.

Thompson, W. L., G. C. White, and C. Gowan. 1998. Monitoring vertebrate populations. Academic Press, San Diego, California.

Urquhart, N. S., and T. M. Kincaid. 1999. Designs for detecting trend from repeated surveys of ecological resources. Journal of Agricultural, Biological and Environmental Statistics 4:404–414.

Urquhart, N. S., S. G. Paulsen, and D. P. Larsen. 1998. Monitoring for policy-relevant regional trends over time. Ecological Applications 8:246–257.

Verdin, K. L., and J. P. Verdin. 1999. A topological system for delineation and codification of the Earth's river basins. Journal of Hydrology 218:1–12.