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FISH INVENTORY and MONITORING TECHNICAL GUIDE for WADEABLE STREAMS on NATIONAL FORESTS

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Executive Summary

This guide addresses inventory and monitoring strategies for fish in wadeable streams. It focuses on monitoring associated with National Forest Management Act planning and is intended to apply primarily to monitoring efforts at the National Forest level scale. Primary topics covered in the guide are key biological, logistical, and statistical issues relating fish surveys, general strategies for fish inventory and monitoring, and sections specific to distribution, abundance, trend, and purposive survey requirements. The guide is intended to be applied on all National Forests and for the majority of fish species likely to be encountered. The authors recognize that there will some site and species specific situations where these standards can not be applied. The guide will be updated every five years to accommodate new information and survey techniques.

The overriding focus of the guide is to encourage the use of randomized and statistically defensible survey standards to assess fish populations at the National Forest scale. Implementation of these standards will allow the USDA Forest Service to demonstrate sustainable development of fisheries resources and meet regulatory and administrative policy goals. A key component of the survey standards that will allow the Forest Service to meet these goals is the use of consistent levels of precision and statistical power when developing survey designs. As such, we have underlined our specific recommendations throughout the report and summarize them here as follows:

1. Randomized survey designs should be used whenever possible and the scope of inference should be increased via coordinated sampling with other agencies and organizations to increase efficiency and decrease survey costs. For large scale surveys (i.e. > 6^{th} order HUC) the generalized random-tessellation stratified design should be implemented to ensure equal sample distribution across the landscape.
2. The minimum length of a sample unit should be 100 m.
3. Trend surveys should use permanent sample sites.
4. Distribution surveys should have a minimum sample size sufficient to detect the presence of a target population within 20% of the true frequency (confidence interval) 80% of the time (power). We recommend that the interim threshold density should be set at 0.1 individuals per sample unit (100 m reach) until regional standards are developed (Hoffmann et al. 2005).
5. Abundance and trend surveys should have a minimum sample size sufficient to detect the abundance of a target population within 20% of the true abundance 80% of the time with $\alpha = 0.10$.
6. Population estimates (i.e. any abundance estimate that is not an index) should use block nets at the upstream and downstream ends of all sample units.

7. Management actions should be considered (i.e. trigger point) when an estimated 20% change in frequency of presence or overall distribution is observed (Vesely et al. 2006).
8. When using electroshocking, seine, underwater (snorkel), plot/quadrat, redd, or minnow trap techniques to survey fish the standardized methods outlined in Appendix 5 of this guide should be used to allow comparisons of data collected in different National Forests.
9. Purposive sampling is useful for preliminary investigations, but can not provide levels of confidence in estimates of distribution or abundance and therefore, should be only be used when statistically defensible information is not required.

An extensive set of appendices are provided with this guide to aid managers and survey coordinators in the design and implementation of fish inventory and monitoring programs. This manual is not intended to replace existing protocols, but instead bring together all the existing information in a concise and usable format designed specifically for fish in wadeable streams. Much of the information in this guide was synthesized from two existing monitoring texts, Elzinga et al. (1998) and Thompson et al. (1998), and readers are encouraged to review these texts for more detail and supporting information.

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1.1 Introduction

1.1.1 Overview

This technical guide provides direction for inventory and monitoring freshwater fishes in wadeable streams. The guide is designed for use by the U.S. Department of Agriculture Forest Service (Forest Service) consistent with national direction, local priorities, and available funding, and also by interested partners and collaborators. We use the same definition of wadeable streams as the Environmental Protection Agency (EPA 2006) which are streams "… small enough to sample without a boat" and streams " … which fall into the 1st through 5th stream order range". Stream order is a measure of stream size and is defined in more detail in the glossary (Appendix 1). When the protocols described in this technical guide are implemented, the resulting data will meet standards of the Data Quality Act and, therefore, will be legally and scientifically defensible and consistent with data collected elsewhere using the same protocols. We use the same expanded definition of the term "protocol" as per Vesely et al. (2006) to include all aspects of the inventory and monitoring process from sampling design, data collection methods, data analysis methods, and reporting.

The technical guide is divided into seven chapters: overview, general fish inventory and monitoring strategies, dichotomous key, distribution surveys, abundance surveys, trend surveys, and purposive (i.e. project specific) surveys. A large number of appendices are also provided including a glossary, examples of capture efficiencies by fish species and sample technique, sample size calculations, random number table, recommend survey techniques and procedures, field survey forms, recommended statistical analyses, and web links to available analysis programs. The technical guide was written for regional monitoring coordinators and their survey teams, forest biologists, and other agencies and organizations interested in fish inventory and monitoring activities. This introductory chapter provides an overview of the technical guide and describes the business needs that motivate the Forest Service to inventory and monitor freshwater fishes in wadeable streams. This chapter also describes the roles and responsibilities of implementing this technical guide and provides the context of fish inventory and monitoring in relation to other federal and state inventory and monitoring programs.

It has been recognized that a standard format for inventory and monitoring protocols of wildlife, fish, and rare plants (WFRP) would benefit Forest Service staff and other users; therefore, this guide will follow the format recommended by Vesely et al. (2006). The content and structure of this protocol draws heavily from previously completed Forest Service protocols for the northern goshawk (Woodbridge and Hargis 2006) and multi-species inventories (Manley et al. 2006). Also, this protocol establishes fish inventory and monitoring approaches that are consistent with the Aquatic Ecology Unit Inventory Technical Guide

(AEUI) (Potyondy et al. In Press?). The AEUI is a parallel effort of the Forest Service to standardize aquatic habitat inventory and monitoring efforts. To this end, when discussing aquatic ecological units (reach, sub-basin, watershed, etc) we will use the terminology and hierarchical classification scheme of Maxwell et al. (1995) unless otherwise stated.

The terms inventory and monitor have been used to refer to a wide range of activities in environmental sciences and we have chosen to use the definitions provided by Thompson et al. (1998). In this guide inventory refers to "gathering baseline information on a species" spatial distribution and abundance …" and monitor refers to "repeated assessment of status [of a population]… within a defined area over a specified time period." We will use the term "survey" when referring to inventory and monitoring processes together as a survey is simply the partial counting of fish or objects within a defined area and time period (Thompson et al. 1998).

1.1.2 Inventory and Monitoring Goals

The emphasis of this guide is to encourage all Forest Service fisheries managers to consider the context of any fish sampling conducted on National Forest Lands and to adopt probabilistic (i.e. randomized) sample designs and standardized survey techniques whenever possible. However, we recognize that site specific sampling will be required to address focused management questions. For example, many forest managers require information on the presence or absence of certain fish species adjacent to and downstream of timber sales that border streams. This guide includes recommendations for this type of purposive sampling as well (often referred to as *representative sampling*). It is also our hope that the recommendations in this guide will allow purposive sampling to be incorporated into larger scale inventory and monitoring efforts, thereby increasing the efficiency of data collection, and increasing the Forest Service's ability to assess fish populations at the regional and forest level scales.

Collecting fish inventory and monitoring data in a consistent way from strategically designed surveys will enable the Forest Service to achieve the following goals at the forest, regional, and national scales:

Forest-scale Goals

- Provide statistically rigorous evaluations of status and change of selected fish populations in wadeable streams.
- Provide a consistent set of data for forest plan revisions.
- Provide a consistent set of data and **metadata** to populate the Natural Resource Information System (NRIS 2005).
- Provide evaluations of proposed forest development and other site specific activities (i.e. timber harvest, road construction, etc.).
- Provide a spatially defined connection between fish population status and trend and AEUI data.

- Provide data that allow for aggregating similar data to describe broader areas.
- Provide data that allow for evaluating fish distributions, abundance, and trend.

Regional-scale Goals

- Describe fish distributions across several forests.
- Describe the range of variability of fish populations for areas larger than a forest.
- Provide assessment of region-wide aquatic **endangered species** status.
- Provide data for broad-scale assessments.

National-scale Goals

- Provide data on specific fish population trend collected in a similar way so information can be combined across forest and regional boundaries.
- Provide a sampling framework to coordinate work with Federal, State, tribal, and private partners.
- Provide needed information for broad-scale assessments, analyses, and decisions of national significance.
- Ensure consistency of aquatic information in NRIS.

1.1.3 Inventory and Monitoring Objectives

The primary objectives of this guide are to provide standardized protocols for 1) designing site, forest, and regional level fish inventory and monitoring surveys for distribution, population estimates, relative abundance, and trend surveys, 2) field sampling of fish, and 3) reporting (including data analysis and data storage). If these protocols are implemented they will allow comparisons of fish populations at multiple spatial scales.

1.2 Background and Business Needs

1.2.1 Background

Fish make up over half of all vertebrate species in the world and are an extremely diverse and taxonomically complicated group (Helfman et al. 1997). In North America there are over 900 native freshwater fish species with the highest species diversity in the southeastern US. Species diversity declines towards the north (due to recent glaciation) and the west (due to preponderance of arid basins) (Hocutt and Wiley 1986, Moyle and Cech 2004). There are two main groups of fish found in freshwater: euyhaline marine fish and obligatory freshwater fishes. Euryhaline marine fish are marine fish that can tolerate freshwater (< 25-30 ppt salt concentration) for extended periods. Survey techniques for euryhaline fishes will not be addressed in this manual. The focus of this manual is on obligatory freshwater fish that inhabit wadeable streams. There are two broad types of obligatory freshwater fish: freshwater dispersants and salt water dispersants. Freshwater dispersants dominate the fish fauna of

North America (> 90% of the species) and are characterized by species for which their distribution can best be explained by dispersal through freshwater habitats (Moyle and Cech 2004). The distribution of salt water dispersants are best explained by dispersal via marine environments (e.g. salmon) and these species often dominate coastal streams.

It is beyond the scope of this technical guide to describe the life history and ecological requirements of each obligatory freshwater fish species, so for the remainder of this guide we will discuss survey procedures as they pertain to assemblages of fish species based on the major stream types they occupy, namely cold water (e.g. sculpins, whitefish, salmonids), cool water (e.g. bass, pike, perch, walleye), and warmwater (e.g. sunfish, catfish, cyprinids, pupfish) or other groupings relevant to survey procedures. Cold, cool, and warmwater stream types are often used by management agencies as a means to group streams and lakes for management (Magnuson et al. 1979). In many cases the techniques required to sample freshwater fishes in wadeable streams are relatively limited, and often one or two techniques are the dominant methods employed to capture most species over large geographic areas. This guide will focus on the most common sample techniques used to capture cold, cool, and warmwater fishes across North America.

1.2.2 Business Needs

The Forest Service is motivated to develop and implement a national fish survey protocol for freshwater fishes in wadeable streams for three basic reasons: 1) existing laws and policies, 2) a growing call for standards and a recognition that current survey strategies are insufficient to determine the distribution, abundance, and trend of fish populations at the regional and national scale (Bonar and Hubert 2002), and 3) the current threatened and endangered status of many freshwater fish (Williams et al. 1989, Loftus and Flather 2000).

Existing Laws and Policies

The Forest Service is mandated by the Forest and Rangeland Renewable Resources Planning Act (FRRPA) of 1974, as amended by the National Forest Management Act (NFMA) of 1976, to develop resource management plans for national forest lands every ten years. The Forest Service is also required to assess and monitor national forest resources (including fish) and periodically report their findings to Congress. There are a large number of other acts that require the Forest Service to provide scientifically defensible fish survey data to establish that proposed activities on National Forest Lands will not negatively impact aquatic resources. These acts include, but are not limited to, the National Environment Protection Act (NEPA - 1969), Multiple Use Sustainable Yield Act (MUSYA - 1960), Clean Water Act (CWA - 1972), and the Endangered Species Act (ESA - 1973). A more detailed review of the laws and regulations affecting forest planning can be found at http://www.fs.fed.us/biology/planning/guide/laws.html.

A number of existing national and international policy initiatives that have been adopted by the Forest Service also motivate the continued integration, standardization, and rigorous assessment of survey protocols. The Committee on the Environment and Natural Resources (CENR) joined a White House effort to develop a framework for integrating environmental monitoring programs to provide more compatible resource information (Figure 1). The framework was designed to address the multiple scales and processes of the environment using existing methods designed to monitor various aspects of the environment in the most effective manner possible (CENR 1997). This manual focuses on survey efforts directed at the National Forest scale which is between index sites and Regional Resource surveys (Figure 1).

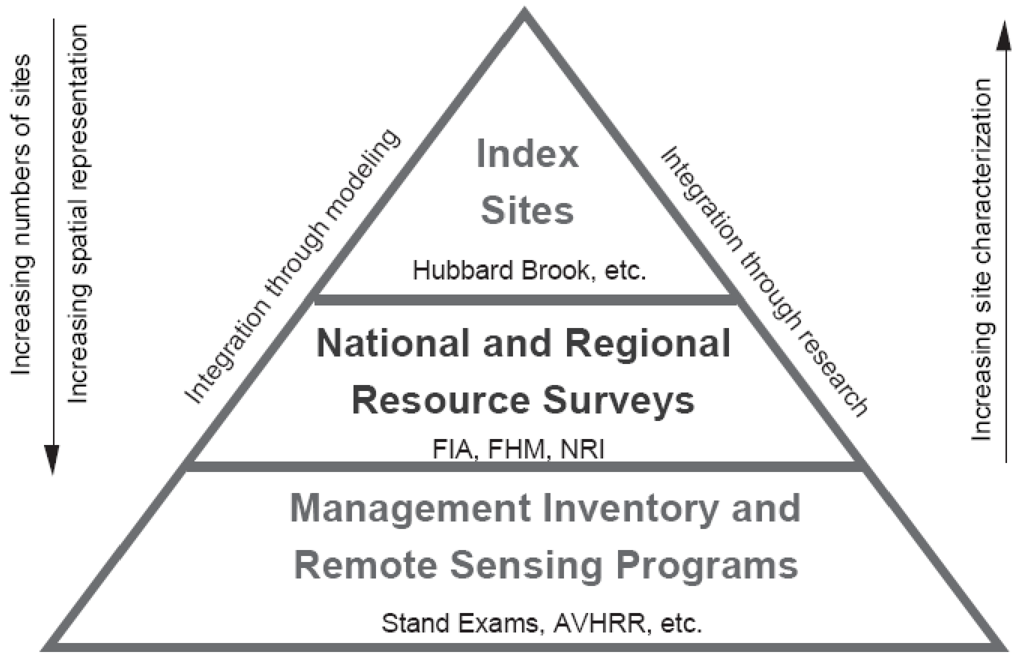


Figure 1. Framework for integrating environmental networks via survey strategies adopted from the CENR (1997). Abbreviations are as follows: FIA = Forest Inventory and Analysis, FHM = Forest Health Monitoring, NRI = National Resources Inventory, AVHRR = Advanced Very High Resolution Radiometer.

Another significant policy adopted by the Forest Service relating to forest sustainability is the Montreal Process Criteria and Indicator (Abee 2000). The Working Group on Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests (Montreal Process) was formed in Geneva in June 1994 to advance the development of internationally agreed criteria and indicators for the conservation and sustainable management of temperate and boreal forests at the national level (http://www.mpci.org/rep-pub/1995/santiago_e.html#c1). The Montreal Process uses six criteria by which sustainable forest management may be assessed. Each criterion is characterized by a set of related indicators which are monitored periodically to assess change. Several of the indicators used in the Montreal Process that relate directly to fish survey include: a) the number of forest dependent species, b) the status (threatened, rare, vulnerable, endangered, or extinct) of forest dependent species at risk of not maintaining viable breeding populations, as determined by legislation or scientific assessment, c) the number of forest dependent species that occupy a small portion of their former range, and d) the population levels of representative species from diverse habitats monitored across their range.

Call for Standards

This technical guide is in part a response to the recognized need for nation wide standards and protocols for inventorying and monitoring fish and wildlife species both within the Forest Service (Manley et al. 2006, Vesely et al. 2006) and other government agencies and organizations (Barbour et al. 1999, Bonar and Hubert 2002, Peck et al. 2006). Recent government accounting procedures have highlighted what has been recognized for some time, that many government agencies do not have the “available, timely, reliable, and complete data” required to effectively determine the status and trend of water quality and aquatic resources (GAO 2000, 2004). The passing of the Data Quality Act (2001) has also increased the scrutiny of agency data and the conclusions that are drawn from them.

Increasingly resource management agencies are being required to provide data that can answer fundamental questions such as “are current management initiatives causing decreases in distribution and abundance of species of management concern”, or “have recent restoration efforts resulted in increases in target species”? To answer these types of questions it is acknowledged that the Forest Service will require standardized survey protocols (Overton et al. 1997, Bonar and Hubert 2002, Henderson et al. 2005). However, district level sampling is still driven more by local objectives and often without statistically rigorous sampling designs. The status and trend of many fish populations can not be determined at the regional and national scale because of the multitude of local protocols in use throughout the country, the inconsistent manner in which many protocols are used, the wide variety of objectives of individual projects, and the lack of stringent quality control measures (e.g. Peterson and Wollrab 1999).

Threats to Populations and Species

Human caused species extinctions are happening at a greater rate than any other pre-historic extinction events (Myers and Knoll 2001), and freshwater species are disproportionately more threatened than terrestrial species (Warren and Burr 1994, Richter et al. 1997, Loftus and Flather 2000). For example, fewer than 20% of terrestrial vertebrates in the United States are at risk of extinction, whereas almost 40% of amphibians and fish, and 67% of unionid mussels are at risk of extinction (Richter et al. 1997). Habitat destruction and competition from introduced species are consistently recognized as the two leading causes of species extinctions in both terrestrial and aquatic ecosystems (Miller et al. 1989, Warren and Burr 1994, Richter et al. 1997, Wilcove et al. 1998).

The US Fish and Wildlife Service (USFWS) currently lists 138 fish species as threatened or endangered which is more than any other vertebrate group (http://ecos.fws.gov/tess_public/Boxscore.do). An additional 13 fishes are listed as candidates for listing and one fish (Oregon populations of coho – *Oncorhynchus kisutch*) are proposed for listing. There are also numerous fish species listed as sensitive or as species of management concern by state agencies (e.g. salmonid species in the western US, minnows and killfish in the southwestern states, and darters in the southeast). The southwestern and southeastern states have the most species listed as endangered, threatened, or of management concern (Williams et al. 1989).

In summary, the Forest Service needs to develop and implement standard protocols for determining the distribution, abundance, and trend of freshwater fish species because:

- These data are required to meet regulatory and policy mandates,
- Some methodologies do not meet rigorous statistical standards required to evaluate status and trend at the regional and national scale,

- New petitions to list various fish species or distinct population segments are inevitable, and the Forest Service will be required to provide information on status and trend of populations in the future, and
- Public and other government agencies will continue to ask the Forest Service for information on the status of various fish species on National Forest lands, because of the current large numbers of threatened and endangered fishes in North America.

1.3 Key Concepts

The issues faced when inventorying and monitoring fish are similar to those of other vertebrate species in that complete counts (i.e. census) are almost never possible, and that the techniques for capturing individuals are not 100% efficient (i.e. some individuals will be present in a sample unit but will go undetected) (Thompson et al. 1998). A specific set of survey design criteria and assumptions will have to be met for fish survey protocols to be reliable and statistically defensible. Confounding these requirements are the individual life history characteristics of each species that can affect their detection rate, capture probability, presence/absence, and behavior. Below we describe the key concepts related to the survey of freshwater fish in wadeable streams which are divided under three main headings: species biology and life history characteristics, habitat types, and statistical issues.

1.3.1 Species Biology and Life History Characteristics

North American freshwater fish display a wide variety of biological adaptations and life history characteristics that can directly and indirectly influence surveys. The two main groups of freshwater fishes, freshwater and salt water dispersants, have many life history characteristics in common and some that are unique to each group. Below we describe some of the main biological characteristics and life history characteristics of some common species groups with examples of how they can influence survey results. We can not stress enough that a comprehensive understanding of each species' biology and life history characteristics are desirable prior to beginning any survey. If the species of interest does not have well documented life history characteristics, it is strongly recommended that a pilot study be initiated prior to formal survey (see Section 2.2.3 for more detail).

Fish Biology

Approximately 41% of all fish species inhabit freshwater despite only 0.01% by volume of the water in the world being available as freshwater (Horn 1972). The relatively large number of fish species present in freshwater environments is due, in part, to the rapid evolution of species as they invaded complex freshwater habitats from the ocean. It is speculated that isolation of species within large watersheds facilitated this rapid evolution of freshwater species (Helfman et al. 1997, Moyle and Cech 2004). Freshwater habitats are also typically more productive than much of the ocean because they are shallower, allowing more light penetration, and subsequent algae and aquatic plant growth (Moyle and Cech 2004).

Behavioral adaptations of fishes to different environments within freshwater directly influence their susceptibility to different sampling procedures. For instance there are a large number of species (e.g. many trout and salmon, Salmonidae) that are mid-water, generalist predators that inhabit cold or cool, clear streams. These species are generally the most susceptible to electrofishing procedures because of the position they take up in the stream and their high activity during the day (visual predators). However, numerous species occupy niches in close proximity to the substrate (e.g. sculpins, Cottidae) or are more active during the night (e.g. catfish, Ictaluridae) which generally reduces their susceptibility to electrofishing. Physical adaptations of fishes can also influence their susceptibility to capture. For example, many sculpins do not posses a swim bladder as an adaptation to living on the bottom of fast moving streams. Without a swimbladder sculpin that are stunned by electrofishing will not float to the surface and possibly go undetected. Competition between species can also influence survey results. For example, even amongst groups of fish that are mid-water, generalist predators, some species occupy dominant or preferred feeding positions compared to less aggressive species (Hearn 1987, Griffith 1988, Grossman et al. 1998). Therefore, a species susceptibility to capture may vary depending on what other species are present in the stream.

Life History Characteristics

The two dominant life history types of freshwater fishes that can influence surveys are freshwater dispersant and salt water dispersant species. Freshwater dispersant species have three main life history strategies: resident, fluvial, and adfluvial. It was presumed until relatively recently that resident life history behavior was common in stream fish, and that fish moved relatively short distances (< 100-500 m) throughout their life time (Gerking 1958). However, recent advances in telemetry and fish marking devices, plus an increased awareness of metapopulation dynamics (see Section 2.2.1 for further description), have challenged the assumptions of limited movement in stream fish (Gowan et al. 1994, Fausch et al. 2002). Understanding the mobility of fish populations, when they move, where they move to, why they are moving, and possible anthropogenic (dams, irrigation canals, etc) and natural (falls, gradient, etc.) barriers to movement are all important considerations when planning and implementing fish surveys.

Salt water dispersants move between fresh and salt water environments during different periods of their life history. In North America salt water dispersants have two basic life history expressions that can influence surveys: *anadromy* and *catadromy*. Anadromous species, such Atlantic and Pacific salmon, are characterized by adult migration from salt water to freshwater habitats for spawning purposes. Juvenile salmon spend as little as a few days to several years rearing in freshwater before migrating to the ocean to feed and mature

(Quinn 2005). Catadromous species in North America are best represented by the eel family Angullidae. Many North American eels spend most of their lives in freshwater only migrating to the ocean to spawn. The primary influences of anadromy or catadromy on fish surveys are their affects on the spatial and temporal distribution of individual fish, age classes, and stocks. Other behavioral characteristics of fish that can significantly influence the results of surveys include fright responses (Ensign et al. 2002, Plachta and Popper 2003), nocturnal activity (Thurow 1996, Young et al. 1997), and schooling (Adams et al. 2004),.

1.3.2 Habitat Attributes

Habitat can be highly heterogeneous within wadeable streams and fish distribution and abundance can be greatly effected by habitat types (Schlosser and Angermeier 1995, Fausch et al. 2002). The ability to capture fish is strongly dependent on habitat type with complex habitats (e.g. LWD, boulders, vegetation) often being difficult to survey accurately. Probably the most significant habitat attribute influencing our ability to inventory and monitor fishes is water clarity. Water clarity is typically measured or described by the degree of turbidity, which is an approximation of the amount of suspended sediment in the water. Electrofishing as typically practiced requires reasonable water clarity in order to net stunned fish that are floating below the surface (Barbour et al. 1999). If the stream bottom can not be seen at depths > 1.5 m the stream is considered moderately turbid to turbid and electrofishing success will bias (underestimation) (RIC 2001).

Other techniques have been employed in turbid streams to increase capture efficiency including hand seines (Patton et al. 2000), electric seines (Bayley et al. 1989), benthic samplers (Weddle and Kessler 1993, Peterson and Rabeni 2001), and various traps designs (Hilderbrand and Kershner 2000, Ketcham et al. 2005a).

Habitat complexity is probably the next most significant variable influencing fish capture efficiency (Habera et al. 1992, Peterson and Dunham 2001). In general the more complex the habitat the more likely fish can evade capture. Complex habitat can include aquatic vegetation, large woody debris, undercut banks, and boulders. Extremely turbulent water can also be difficult to sample because it is usually shallow and fast flowing and it is hard to detect stunned fish.

1.3.3 Statistical Issues

This section deals with some of critical statistical issues that need to be considered as a result of the fish and fish habitat attributes discussed above. First we describe define a few basic terms that we use to consistently describe statistical design and sampling issues. These definitions, unless stated other wise, are based on those described in Thompson et al. (1998) and Vesely et al. (2006). We then describe issues related to sources of variance and validation of estimates.

The hierarchy of sampling terms goes from an individual element to target population (Table 1). In fisheries sampling an *element* is usually the individual fish that is measured or enumerated. However, an element can be any individual, object or item of interest that is directly measured, counted, or recorded. A *sampling unit* is typically a defined length or area of stream (i.e. a reach) or a specific habitat type (i.e. riffles, pools, etc.) where individual elements are captured and/or counted. A *sampling frame* is a collection of all possible sample units within the area of study. When deciding on an inventory or monitoring project it is important to decide what life stage, species, and/or group of species you wish to inventory. This is referred to as the *target population*. A *target population* is a group of elements within a defined area and time period.

Table 1. The hierarchy of sample design terms going from the most basic (element) to the most general (Target Population). Adapted from Thompson et al. (1998).

Increasing Unit Size -->					
Term	Elements	Sample Unit	Sample Frame	Sample Population (Sample Size)	Target Population
Example	Individual Fish	100 m sample reach	All 100 m sample reaches in a watershed	Number of 100 m reaches surveyed	All rainbow trout ≥ 200 mm in watershed X

The goal of “regionally sponsored” surveys should be to collect data that can be used to make inferences about the status and trend of fish populations beyond the individual sites that are sampled, and to collect data that can be compared to other fish surveys at the regional and national level by way of the use of standard protocols. S*tatistical* inference (also called *scope of inference*) is a key concept for all statistically defensible surveys. Statistical inference refers to the space and time over which the results of the survey can be extrapolated(Vesely et al. 2006). If the choice of sample sites is not random, then the statistical inference of a survey is limited to *only* those sites that were sampled. Therefore, it is essential that random sampling techniques be employed if the project objectives are to use the survey results to draw conclusions about sites that were not sampled.

Therefore, if you wanted to know the relative abundance of smelly darter (*Aroma noxious*) > 100 mm in the Stinky watershed in the summer of 2007, and you sampled the drainage by randomly selecting 5% of all the 100 m reaches delineated on 1:24,000 topographic maps, you would define each of the following thus:

Element = each smelly darter > 100 mm

Sample Unit = a 100 m randomly spaced reach

Sample (size) = the number of sample units that are actually measured in the field (the number of 100 m reaches that fish are collected from to make inferences about the entire watershed). In most cases the number of samples should be determined by first determining the level of precision required (see precision below).

Sample Frame = all 100 m sample reaches within the Stinky watershed

Target Population = all smelly darter > 100 mm within the watershed during the summer of 2007.

Scope of Inference = the entire Stinky watershed during the summer of 2007.

This type of sampling would produce a population estimate of the number of smelly darters > 100 mm in the Stinky watershed. The population estimate is known as a parameter estimate. There are statistical methods to estimate how accurately parameter estimates likely reflect the true population but which can never be known (see below).

The above example assumes that a probabilistic (i.e. randomized) survey design was employed. If, however, the survey project was designed for a site specific project the definitions would change. For example, if we wanted to know if smelly darters were present, but only as they relate to a proposed road crossing in a Forest District. The objective of the study may be to determine if smelly darters are present during the proposed construction period of July 1 to August 31. If the darters are present, then special sediment control measures will have to be implemented, if the darters are absent, no control measures will be required. For this project we would define the statistical terms thus:

Element = all smelly darter in any age/size class

Sample Unit = a 100 m reach purposively located upstream and downstream of the proposed road crossing.

Sample (size) = two reaches, one upstream and one downstream of the road crossing. The level of precision of purposive sampling in not calculated.

Sample Frame = the two reaches upstream and downstream of the proposed road crossing.

Target Population = all smelly darter 100 mm within the two selected reaches during the proposed construction period of the road crossing (July 1 to August 31).

Scope of Inference = the two reaches adjacent to the proposed road crossing.

There are a few other key terms used frequently when describing sampling populations including: alpha, confidence interval, power, and threshold density. We provide brief definitions of these terms below.

Alpha value (α) = the probability of rejecting the null hypothesis when it is actually true. It is also referred to as the Type I error rate. Traditionally more emphasis was placed on avoiding Type I errors than Type II errors (see below). The *α* level is usually set at 0.05. An *α* level 0.05 means we are willing to accept a $\leq$ 5% chance of making a Type I error (i.e. rejecting the null hypothesis of no effect when it was true). The α value is also used to set the width of confidence intervals on parameter estimates, such as the number of fish per reach or m^2

(see confidence intervals below). Increasingly in environmental sciences more emphasis is being placed on avoiding Type II error rates by increasing the *power* of surveys.

Power (1- ß) = statistical power refers to the probability of being able to detect an effect (i.e. reject the null hypothesis) if indeed there is an effect. If we conclude that there was no effect when there is an effect (i.e. pollution at a site decreased the abundance of species X, but we failed to detect it) that is termed a Type II statistical error. Most scientific experiments are designed to avoid a Type I statistical error which in this case is concluding that pollution decreased the abundance of species X when there really was no effect. The term power is also used to describe the ability of a survey to detect the presence of a minimum density of the target population. For example, if the target species is rare, it is likely that it will not be detected at some sites even when it is in fact present. If we want to be relatively confident in our estimates about the presence/absence of a rare species then we should design a survey with relatively high levels of power. Many researchers use 80% power as a minimum level. Power is a very important concept in the design of surveys and biologists should make sure they are comfortable with the concept before designing a survey. For more detailed discussions on power see (Peterman 1989 , Peterman 1990, Peterman and M'Gonigle 1992, Taylor and Gerrodette 1993, Thomas and Krebs 1997).

Confidence Interval = an estimate of precision around a sample mean, sample proportion, or a population estimate which indicates the likelihood that the interval includes the true value (Elzinga et al. 2001). For example, if you want to get a population estimate of species X in two watersheds we might sample five 100 m reaches (sample units) out of a possible 100 reaches in each watershed (i.e. sample size is 5% of the sample frame). If the mean number of fish per reach in each watershed is 9 fish, then by multiplying 9 fish by the possible 100 reaches we get a population estimate of 900 fish in each watershed (Table 2). We calculate the confidence intervals for these population estimates to determine how precise these population estimates are. To calculate a confidence interval we need to calculate standard deviation and standard error first. Most common spreadsheet and statistical packages will calculate these values for a series of samples (e.g. see “Tools”, “Data Analysis”, and “Summary Statistics” in Excel). We also have to decide the confidence interval level to use in the calculation of the confidence interval. The confidence interval level is the probability that the confidence interval width contains the true value. Often the level is set at the familiar 95% level which relates to us being 95% confident that our population estimate will contain the true number of fish in either watershed 95 times out of 100. The last value we need is an appropriate t-value. A t-value is typically derived from statistical tables and by looking up the confidence level interval (in this case 95%), and the degrees of freedom (n – 1). The following steps were used to calculate the 95% confidence intervals for our two population estimates in watershed A and B:

- Calculate the standard deviation (SD) of our five reach samples as

$$SD = \sqrt{\frac{\sum (x_i - \bar{x})^2}{n-1}}$$

where SD = standard deviation, n = number of samples (5), x_i = number of fish in reach $_i$, $\bar{x}$ = the mean

- Calculate standard error (SE) as

$$SE = SD / \sqrt{n}$$

- Determine the appropriate t-value for 95% confidence intervals (α = 0.05, degrees of freedom = n-1 or 5-1 = 4) in this case 2.776 (see Zar 1984 or any other standard statistical text for t-tables)
- One half the 95% CI = SE * t-value
- Full 95% interval = population estimate ± one half of 95% CI

In table 2 it can be seen that the full interval for watershed A is much smaller (666-1134) compared to watershed B (-129-1929). Both these ranges indicate that we expect the true number of fish in each watershed to fall within these ranges 95% of the time. Obviously the estimate for watershed B is not very useful because the interval is so large that it has negative values.

Table 2. Example of calculating 95% confidence intervals for two population estimates based on five samples from 100 possible reaches. Abbreviations are N = number of reaches sampled, SD = standard deviation, SE = standard error, Pop Est = population estimate in each watershed, t-value = 95% CI = 95%

confidence interval, Lower = lower bound of 95% interval, Upper = upper bound of 95% confidence interval.

	No. Fish Per Reach													
Watershed	**1**	**2**	**3**	**4**	**5**	**Mean**	**N**	**SD**	**SE**	**Pop Est**	**t-value**	**95%CI**	**Lower**	**Upper**
A	8	11	8	11	7	9	5	1.9	0.8	900	2.8	2.3	666	1134
B	15	15	0	0	15	9	5	8.2	3.7	900	2.8	10.3	-129	1929

Threshold Density = the assumed lowest densities the target population is likely to be found at within a sample unit. The more rare the species is, the more sampling effort that is required to maintain a certain level of power. For example, if a species is present at a density of 100 individuals per 100 m of stream far fewer reaches will need to be sampled to have an 80% probability (power) of detecting its presence compared to a species that has a density of 0.1 individuals per 100 m of stream.

The main issue threshold density raises for survey designs is how rare does a species have to be before we are willing to consider it “absent”. Obviously this is site, species, and jurisdictionally specific. The general idea is that setting a threshold density implies that populations below the minimum density are below an hypothesized minimum viable population (Rieman and McIntyre 1995). Peterson et al. (2002) and Hoffmann et al. (2005) discuss these issues in detail with differing conclusions. For consistency, we recommend following Green and Young (1993) that suggest that a species be considered rare at densities below 0.1 individuals per sample unit (sample unit = 100 m stream reach). Therefore, when calculating sample sizes a threshold density of 0.1 should be used and if the species is not detected it can be considered to have a density at or below 0.1 within the sample frame if all sampling assumptions are met. It is clear that the issue of threshold density requires more input from policy makers and resource managers at the regional level and this recommendation should be adjusted as more information becomes available (Hoffman et al. 2005). We discuss specific recommendations for the application of power and threshold density for distribution surveys in more detail in Section 4.0.

Sources of Variance

Fish populations naturally fluctuate due to a variety of factors including flow (Mueller et al. 2003, Albanese et al. 2004), season (Muhlfeld et al. 2001, Colyer 2002), water temperature (Bjornn 1971), and natural disturbance (Roghair et al. 2002, Dunham et al. 2003, Magoulick and Kobza 2003, Dodds et al. 2004). This natural variability makes detecting changes in abundance difficult, especially when other sources of variance due to survey design and sample methodologies are large. It is therefore, the goal of any survey protocol to reduce as much as possible the sources of variance associated with survey design and sampling methods.

Validation of Estimates

It is recognized that no sampling technique, with the possible exception of treatments with ichthyocide or some other destructive sampling, are 100% efficient (Boccardy and Cooper 1963, Jacobs and Swink 1982, Metzger and

Shafland 1986). Therefore, whenever a sampling technique is employed, fewer fish are counted than actually occur in the sample unit (i.e. negative bias). For example, Thurow et al. (2006) determined that snorkel estimates of bull trout abundance in a small stream were at best 33% of the true abundance. Thurow et al. (2006) estimated the true abundance using a mark recapture estimate prior to conducting the snorkel survey. Mark recapture techniques are recognized as the most precise method available in most situations (assuming all the assumptions are met) for obtaining estimates that are as close as possible to the true abundance (Peterson and Cederholm 1984, Peterson et al. 2004, Rosenberger and Dunham 2005). Therefore, when designing surveys for estimating a species abundance (i.e. population estimate) it is necessary to either use methods that adjust for incomplete capture probabilities (e.g. mark recapture), or use relative abundance estimates with double sampling procedures (Eberhardt and Simmons 1987). Using two relative abundance methods will provide estimates of capture efficiency and sampling bias (see chapter 4 for more detail).

1.4 Roles and Responsibilities

1.4.1 National Responsibilities

- Lead and facilitate service-wide, interdisciplinary development of fish surveys protocol. Coordinate with other Forest Service survey protocol groups and initiatives.

- Coordinate with other agencies on aquatic inventory, monitoring, and classification standards. Ensure that fish survey protocol is consistent with standards adopted by the Federal Geographic Data Committee (FGDC).
- Implement fish survey protocol as part of the Forest Service inventory and monitoring framework.
- Support and evaluate regional implementation of a fish survey protocol as scheduled in strategic inventory plans. Validate compliance with national survey standards.
- Manage the change management process in a timely way.
- Participate on boards that guide the design and implementation of NRIS and core GIS requirements to ensure that results of NRIS and GIS actions support the fish survey protocol.
- Ensure consistency of fish survey protocol data by developing a field-personnel training program.

1.4.2 Regional

- Guide regional implementation of the fish survey protocol to follow national standards and protocols.
- Develop regional attributes and protocols as needed.
- Coordinate with internal and external regional entities and neighboring Forest Service regions to stimulate collaboration of work and correlation of results. This process includes implementation and adoption of regional guidance on a fish survey protocol, NRIS, and GIS.

- Coordinate with States and tribes on fish data collection as needed.
- Participate on boards that guide the design and implementation of NRIS and core GIS requirements to ensure that results of NRIS and GIS actions support fish survey protocol.
- Ensure performance measures and outcomes are accomplishing fish survey work. Conduct training programs for field personnel to ensure fish survey protocol data consistency. Institute interdisciplinary field reviews of fish survey protocol products for consistency and quality.

1.4.3 Forest

- Conduct fish surveys to meet forest needs on schedule and within budget.
- Recommend budgeting and scheduling of fish survey work to fulfill information needs.
- Coordinate with internal and external local entities, neighboring Forest Service administrative units, and the regional office to stimulate collaboration of work and correlation of results.
- Integrate fish survey protocol products in forest planning and forest- and project-level assessments, as applicable.

1.5 Relationships to Other Federal Inventory and Monitoring Programs

1.5.1 Forest Service Programs

The Forest Service recently recognized that effective inventory and monitoring of forest resources is essential for accomplishing the mission of the Forest Service. Therefore, the Ecosystem Management Corporate Team and the Interregional Ecosystem Management Coordinating Group (EMCT/IREMCG) agreed in 1999 to charter an agency-wide task team comprised of National Forest System (NFS), Research and Development (R&D), and State and Private Forestry (S&PF) representatives from the Washington Office (WO), Research Station (Station), and Regional Office (RO) levels, as well as key external partners to coordinate future survey efforts (http://www.fs.fed.us/emc/rig/iim/).

There are numerous large scale fish and aquatic habitat driven survey initiatives that have been underway for several years in the Pacific Northwest (Reeves et al. 2003) and the southwest (??). The PACFISH/INFISH Biological Opinion (PIBO) Effectiveness Monitoring Program was initiated in 1998 to provide a consistent framework for monitoring aquatic and riparian resources on most Forest Service and Bureau of Land Management lands within the Upper Columbia River Basin (Henderson et al. 2005). The Aquatic and Riparian Effectiveness Monitoring Program (AREMP) has also developed a set of standardized sampling fish habitat protocols as part of the Northwest Forest Plan (Reeves et al. 2003).

This guide does not address inventory and monitoring of fish habitat and instead will rely on the Aquatic Ecology Unit Inventory Guide (AEUI) (Potyondy et al. in press) for the collection and analysis of all stream habitat data. The AUEI is a synthesis previous fish habitat protocols and "… provides data standards and protocols necessary for getting basic physical, chemical, and biological information to describe status and trend of aquatic systems at the valley segment and river reach levels" (Potyondy et al. in press).

The Forest Service has also developed a technical guide called the Multiple Species Inventory and Monitoring (MSIM) protocol. The MSIM provides a framework for collecting presence/absence data on a variety of vertebrate species, including fish, over broad spatial extents. Our guide is complementary to the MSIM because it has a similar monitoring objective, but is expanded to include population estimation and does not use the FIA grid to allocate sampling effort.

The Forest Service has also initiated the coordination of data collection and storage with the formation of the National Inventory and Monitoring Action Plan (2000) and the NRIS, which is over seen by the Ecosystem Management

Coordination staff. All data collected using this manual will be compatible with the NRIS.

1.5.2 Programs in Other State and Federal Agencies

There are a wide variety of other government agencies actively involved in survey of fish and fish habitat. The EPA has probably the most extensive monitoring system in place for wadeable streams at the national level (Barbour et al. 1999, EPA 2004, 2006). The Forest Service will use many of the same survey design and sample methods that have been field tested by the EPA in an effort to make the data compatible.

There are also a number of large scale monitoring efforts in the Pacific Northwest that are all focused on trying to standardize fish survey methods. These efforts include the Pacific Northwest Aquatic Monitoring Partnership (PNAMP), Integrated Status and Effectiveness Monitoring Project (ISEMP), and Collaborative Systemwide Monitoring and Evaluation Project (CSMEP). These methods all focus on salmonids. A recent salmonid inventory and monitoring protocol was developed as part of the PNAMP process (Johnson et al. 2007).

1.6 Quality Control and Assurance

This manual is a synthesis of many existing stream inventory protocols that have been developed by federal, state, and provincial agencies (see a large collection of existing fish survey manuals compiled by S. Bonar at the following site http://www.ag.arizona.edu/srnr/research/coop/azfwru/scott/ under “links” and

“Existing State/Provincial Standard Sampling Protocols for Freshwater Fishes”). We have also tried to include information from a large number of studies conducted recently attempting to address some fundamental issues regarding fish inventory and monitoring techniques (Thurow 1994, Peterson and Rabeni 1995, Thompson et al. 1998, Dunham and Davis 2001, Peterson et al. 2004, Peterson et al. 2005, Rosenberger and Dunham 2005). This protocol was developed with the use of these and many other primary literature sources. The protocol was also evaluated (statistical and ecological review) internally within the Forest Service.

1.7 Change Management

This technical guide is considered a draft until the survey designs has been implemented for at least one year in at least one region. After the first year, we anticipate revising the manual as needed based on comments received from people implementing the manual. The Data Storage section will be expanded to describe in detail the structure of the national database and the data fields that will be routinely migrated to NRIS Fauna. Guidelines for constructing a field data entry form might be revised for better efficiency and/or clarity. A website will be developed where updates and additional tools and information can be located for easy retrieval. This manual will be reviewed 5 years after the first year’s revision to determine if additional changes are warranted.

Chapter 2.0 General Strategies for Fish Inventory and Monitoring Programs

2.1 General Strategies Introduction

This chapter outlines the planning, design, data collection, data storage, data analysis, and reporting recommended for all types of fish surveys in wadeable streams. We have chosen to stray slightly from the recommended outline for Forest Service inventory and monitoring manuals by Vesely et al. (2006) because fish inventory and monitoring projects can often have very different objectives (e.g. presence/absence versus population estimate) yet the field techniques may be very similar (e.g. electrofishing can be used for distribution, abundance, and trend monitoring). Therefore, this chapter will summarize all the assumptions, theory, and procedures that are common to all fish inventories regardless of the objectives, and the following chapters will deal with the specific requirements of the three main objectives covered in this manual, namely distribution (Chapter 3), abundance (Chapter 4), and trend (Chapter 5) surveys. Chapter 6 provides guidance for purposive sampling.

2.2 Planning and Design

The planning and design of fish survey strategies in wadeable streams is obviously complicated by the fact that this manual is designed to be applicable for multiple species across all National Forest lands. The large number of fish species and associated life history strategies makes it impossible to define a detailed conceptual model for each species or even species group. However, there have been some conceptual models developed recently that highlight some common life history traits of many stream dwelling (lotic) fish and the major anthropogenic and natural stressors that shape their distribution and abundance (Schlosser 1991, Fausch and Young 1995, Schlosser and Angermeier 1995, Rieman and Dunham 2000, Fausch et al. 2002). The following section draws heavily on these proposed models as a way of outlining an overall conceptual model for lotic fish that is relevant to the development of survey strategies. We stress however, that a thorough understanding of the target species life history characteristics is required to recognize if this general model is valid for each specific situation.

2.2.1 Freshwater Fish Life History and Conceptual Model

Conceptual models are used to highlight ecosystem components and processes that are hypothesized to affect the distribution and abundance of target species, as well as any potential information gaps (Manley et al. 2000). The key driver of lotic systems is the geoclimatic setting of the watershed which controls stream patterns by affecting the magnitude and frequency of terrestrial and aquatic disturbance events (Maxwell 1995). The distribution and abundance of fish throughout space and time are primarily influenced by spatially and temporally diverse disturbance events along with hydrologic processes that act on stream patterns and dictate the presence and arrangement of aquatic habitat types

(Vannote et al. 1980, Schlosser and Angermeier 1995, Dunham et al. 2003, Magoulick and Kobza 2003, Townsend et al. 2003).

Many lotic fish require a complex arrangement of habitat types at varying levels of spatial and temporal scales. Schlosser (1991) and others have attempted to describe these relationships with a simple conceptual model of fish life cycle requirements and the associated movement between habitat types. The model shows three basic life history stages common to most lotic fish: spawning and egg rearing, feeding and growth, and refugia (Figure 2). The important things to understand about this model is that the physical connections between the habitats are critical and that the habitats themselves often exhibit a large degree of spatial and temporal heterogeneity within and between watersheds.

The importance of this spatial and temporal environmental heterogeneity, and the importance of movement between habitat patches, has been demonstrated by numerous studies (Gowan et al. 1994, Fausch et al. 2002, Rodriguez 2002, Vokoun and Rabeni 2005, Quist et al. 2006b). Infrequent movement of individuals between populations has been termed metapopulation dynamics (Policansky and Magnuson 1998, Gotelli and Taylor 1999). The key concept of metapopulations that is particularly important to fish surveys is that individual populations within a metapopulation are subject to local extinctions and subsequent recolonizations (Gotelli and Taylor 1999). Therefore the health of the entire metapopulation is contingent on dispersal of individuals between populations.

Stressors that affect fish distribution and abundance are processes (natural and anthropogenic) that impact the creation and distribution of aquatic habitats and the connections between habitats and populations. For instance natural wildfires can dramatically change the frequency of aquatic habitats over large areas (watersheds) and in some cases cause the local extinction of fish species (Gresswell 1999, Dunham et al. 2003). The impact of anthropogenic disturbances, such as forest harvesting and livestock grazing, on fish habitat quality and quantity have also been well documented (Meehan 1991).

Fish are an important component of aquatic ecosystems and may require monitoring for a variety of reasons. For instance, the relationship between habitat quantity and quality and fish abundance is not well understood for many species and therefore monitoring habitat may not be an accurate way to determine if management activities are protecting fish populations. Also, habitat quality or quantity may not be the limiting factor for some species. There may be physical barriers preventing them from accessing critical habitats (dams, culverts, etc) or there may be interaction with other non-native species that are preventing them from occupying traditional habitat.

Directly monitoring fish distribution and abundance can be a useful way to monitor overall environmental health because fish (Barbour et al. 1999):

- Are relatively long-lived and mobile,
- Assemblages generally represent a variety of trophic levels (omnivores, herbivores, insectivores, planktivores, piscivores),
- relatively easy to collect and identify to the species level,
- Environmental requirements are comparatively well known for common species,
- Life history information is extensive for many species, and
- Fish account for nearly half of the endangered vertebrate species and subspecies in the United States (Warren and Burr 1994).

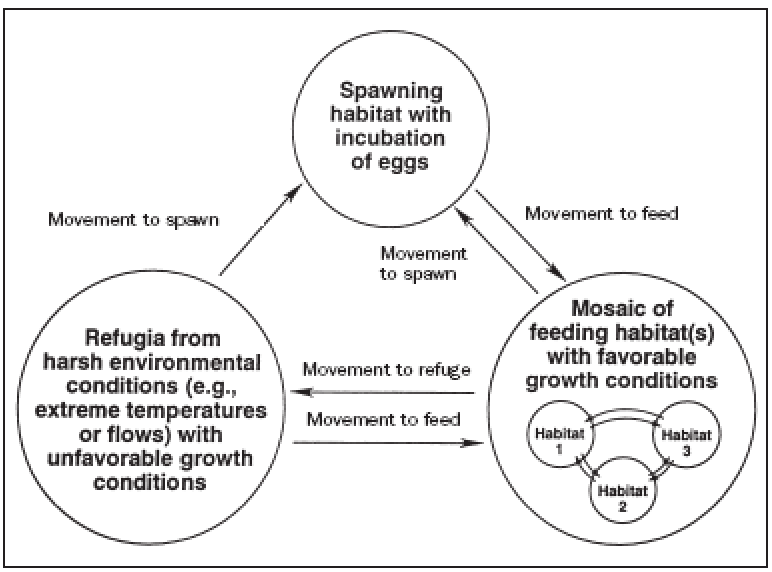


Figure 2. Conceptual model of the life cycle of many lotic fish species showing the connections between the habitat types (Schlosser and Angermeier 1995).

2.2.2 General Fish Inventory and Monitoring Sampling Design

It is important to preface this section with a caution that an adequate sampling design can only be created if the researcher has a good understanding of the life history characteristics of the target species (Thompson et al. 1998, Vesely et al. 2006). The next step is to clearly define the target population.

Target Population

The target population will vary widely depending on local objectives and regional species assemblages. The more clearly the target population is defined the more precisely the survey design can be tailored and the less ambiguity there will be in how the results of the survey are interpreted. The target population is a group of elements (individuals, objects, or items of interest that are measured, counted, or recorded) representing the species of interest within some defined area and time period. The following is an example of a target population definition: summer resident, juvenile and adult Johnny darters in wadeable streams within the Plains River watershed.

<u>We recommend defining target populations based the following criteria: age class, season of residence or occurrence, and migratory behavior (resident, fluvial, adfluvial).</u>

Sampling Frame and Statistical Scope of Inference

Once the particular attributes of the target population have been defined, it is then necessary to define the sampling frame. The sampling frame should include the entire area that the researcher wishes to make statistical inferences about (i.e. draw conclusions about fish presence or abundance). Therefore, if we wanted to know if the Johnny darter was present in the fictitious Plains River watershed (and no other watersheds) then the entire Plains River watershed would be the sample frame. Once a sampling frame is selected it is the only area statistical inference can be made for. Other watersheds must be included in the initial sampling frame and samples must be allocated randomly within them if inferences are to be made (Elzinga et al. 2001).

Scale, both temporal and spatial, is an important issue related to sample frame definition and the scope of inference a study can have. The objective(s) of the study will not be met if the spatial and temporal scale of the sampling frame are inappropriate (Schneider 1994, Johnson and Gage 1997, Wiley et al. 1997). When choosing the size of a sampling frame it is important to determine the scale of the study to the objective of (Schneider 1994, Johnson and Gage 1997, Wiley et al. 1997). An example of matching the spatial scale of a study to the objectives would be the potential difference between the home-range and spatial distribution of resident and adfluvial populations of the same species. An example of matching the temporal scale of a study to the objectives would be the cyclic, two year pattern of pink salmon abundances (Bonar et al. 1989). See Fausch et al. (2002) for a review of scale issues in fisheries management.

Coordination with other agencies and organizations is one potential way to broaden the scope of inference of a study and potentially reduce costs associated with sampling. If a species distribution overlaps administrative boundaries it may be more efficient and biologically meaningful to develop sampling designs in cooperation with other agencies.

We recommend that when choosing sampling frames they be related to the spatial and temporal scale of the objectives of the study, and that survey designs be developed in coordination with other forests, regions, agencies to increase the scope of statistical inference that can be applied to the results.

Sample Selection and Stratification Methods

Once the sample frame has been selected it is then necessary to use existing life history information available for the target population/species to decide how to allocate samples. There are several non-random sampling techniques that have been used in fish and wildlife monitoring including purposive (representative), haphazard, and convenience sampling (Thompson et al. 1998). We strongly discouraged the use of any of these techniques for the purposes of inventory and monitoring fishes in wadeable streams because they do not allow for statistical inference beyond the immediate sample. Two sample allocation techniques that are commonly used are simple random (probabilistic) and stratified random sampling designs units (Thompson et al. 1998, Elzinga et al. 2001). See Vesely et al. (2006) for a description of more complicated approaches such as systematic and adaptive cluster sampling.

Simple random sampling, whereby each sample unit within a sample frame has an equal chance of being selected, is generally acceptable when the sample frame is relatively small and there is little evidence that the target population has a very clumped distribution (Elzinga et al. 2001). Simple random sampling is not recommended if the target population has a strong clumped distribution and/or when budget constraints limit sample sizes because it may not adequately represent the variability on the landscape (i.e. under-represent rare habitats or species) (Vesely et al. 2006). Typically simple random sampling without replacement is less bias than sampling with replacement. Replacement refers to whether a sample is excluded from the selection process once it has been selected once, or whether it can be selected more than once (i.e. re-sampled).

Stratified random sampling is appropriate when there is strong evidence that the target population has a clumped distribution. Stratified random sampling is designed to divide the sample frame into strata based on some pre-existing information on the distribution of the target population (i.e. found in streams with < 25 °C summer average water temperature). The purpose of stratifying the sample frame is to reduce the variation within strata (Thompson et al. 1998, Krebs 1999). Many fish populations tend to have a clumped distribution which almost universally leads to high variation among sample units (reviewed in

Thompson et al. 1998). If the target species is known to be restricted to streams with a mean summer temperature < 25 °C, it would be appropriate to stratify the sample frame into streams with mean summer temperatures above and below 25 °C. Habitat is another common variable used to stratify sample frames as many species have relatively specific habitat requirements. If the Johnny darter was known to prefer sand substrate in shallow (< 20 cm) slow moving streams, it would be appropriate to stratify streams within the sample frame into shallow slow moving streams and all other habitat. Stratification can be much more complex if there is sufficient information on a species life history and behavior.

When choosing strata it should be recognized that strata should remain fixed on the landscape overtime and data collected under one stratification scheme should not be reclassified under another scheme at a later date (Vesely et al. 2006). If researchers believe that specific habitat or stream types have no potential to support a particular target population they can completely exclude them from the sample design, but this should be done with great caution and the understanding that no inference can be made about the target population in areas that are not sampled (Thompson et al. 1998). Although stratified random can increase the efficiency of a sampling design, it does require that the sample frame be stratified prior to the survey which can increase costs substantially.

Another new approach to sample unit selection is the generalized random-tessellation stratified design (GRTS) (Stevens and Olsen 2004). This sampling design is particularly appropriate for large scale linear shaped sample frames like streams which are essentially “one-dimensional continua embedded in two-dimensional space” (Stevens and Olsen 2004). This sampling design preserves the spatial distribution of sample units on the landscape by sequentially numbering all the possible sample points and placing them in order along an imaginary line. Each sample unit (i.e. 100 m stream reach) within a watershed can be placed on the line according to its position within the watershed using stream order or magnitude for example. Points can then be randomly sampled from this line and different inclusion probabilities can be applied depending on the objectives of the study. For example higher inclusion probabilities (the probability that a sample unit will be selected) can be applied to higher order streams and in sub-basins with particular geologic features if existing data suggests these sites are preferred by the target population. Another advantage of the GRTS is that the sample design can be easily updated as priorities change and complications arise in implementing the design (i.e. site access and jurisdictional issues). The GRTS requires GIS or some other similar spatially explicit mapping program to implement and the reader should consult a biometrician and the following resources if they are going to use this method: (Stevens and Olsen 1999, Stevens and Olsen 2004).

For trend surveys a special consideration regarding sample unit allocation is deciding whether to use permanent or temporary sample units. With temporary sample units a set of sample units are randomly selected each sample period and the sample units are independent between sample periods (Elzinga et al. 1998; Thompson et al. 1998). However, permanent sample units are only randomly selected during the first sample period, permanently marked in some fashion, and then revisited in subsequent sample periods.

In general, permanent sample units allow a study design to use less samples to detect changes of a particular magnitude between periods (Elzinga et al. 1998, Elzinga et al. 2001, Roper et al. 2003, Quist et al. 2006a). Permanent sample units will be particularly useful if they can be accurately relocated and if measurements between sample periods are highly correlated. Permanent plots can also reduce the costs associated with finding and traveling to sample sites each year because the logistics are usually worked out after the first sample period. The disadvantages of permanent sample plots are that they are relatively expensive to establish and maintain, can be difficult to relocate, marking can influence the site (i.e. attract or repel target species), sites can be degraded with multiple visits, and changes in habitat conditions can decrease correlation of measurements between sample periods. For example, if a stream is highly dynamic and the channel is constantly moving because of large floods or debris flows, permanent sites may no longer be within the stream channel, or the habitat may have completely changed.

We recommend that randomization of samples be used regardless of the type of design employed (simple, stratified, etc.). Random allocation of samples within a sample frame is recommended because it will allow for evaluation of status and change of fish distribution in wadeable rivers across the forest, regional, and national scales. For small scale surveys (i.e. 7^{th} order HUC and smaller) that are preliminary in nature, we recommend either simple random sampling or stratified random sampling be used in most cases. Simple random sampling should be used when the target population does not have a strongly clumped distribution, or when little is known about its distribution. When the target population has a strongly clumped distribution some type of stratification of the sampling frame is recommended to reduce the variance between samples. We recommend that GRTS sampling designs be implemented for larger scale distribution surveys (i.e. > 6^{th} order HUC) to ensure equal distribution of sample units across the landscape.

We recommend using permanent plots for trend.

Size, Shape, and Spacing of Sample Units

The size of the sample unit used for fish surveys in the past can be categorized as either fixed or variable. Fixed sample unit sizes typically used were 50-100 m units of stream length. Sample units of 50-100 m were chosen because they generally work well for detecting most of the species present, they can be easily sampled in one day (with or without habitat sampling), and they are easy to replicate from site to site. For example, Patton et al. (2000) used 50 m sample units to survey stream reaches in small (2-11 m) wide streams in the Missouri River drainage in eastern Wyoming. They found that sampling three 50 m units using backpack electrofishing were required to detect 90% of the species present, and four 50 m units captured 100% of the species present.

Variable sample sizes generally involve using a sample unit length relative to the width of the sample stream. For example, 20-30 bankfull widths has often been recommended as an appropriate sample unit size for distribution surveys. Other researchers have shown that even larger sample units are required (58-135 stream widths), particularly for rare fish that are widely distributed and only one sample method is used (Paller 1995). The width of stream is usually defined as the mean wetted width, channel width, or bankfull width. Other variable sample unit sizes that have been used include some minimum number of habitat units. For example, on larger streams it has been recommended that each sample unit contain at least two complete meander cycles (i.e. two pool and two riffle sequences) (Peck et al. 2006). However, stream widths can be difficult to measure accurately in the field even with significant effort placed on crew training (Whitacre et al. In Press).

<u>We recommend using a minimum standard of 100 m per sample unit for all fish surveys. This approach will avoid confusion in the field measuring stream widths, apply a consistent approach across all forests, and maintain a reasonable level</u>

<u>of sampling precision. If block nets are used (i.e. for population estimates) they should be placed in an appropriate location outside the sample unit to maintain the full 100 m sample length.</u>

Methods to Control and Measure Observer Bias

Virtually all fish survey methods have some type of observer bias resulting from issues relating to the capture efficiency of the survey method. For example, observer bias can occur when conducting snorkel surveys and individual fish are improperly identified to species or are recorded in the wrong size class (Thurow 1994, Dolloff et al. 1996). Bias associated with capture efficiency has received a considerable amount of study in recent years and it is well recognized that capture efficiency varies among different species, and size and age classes within species (Peterson 1999, Bayley and Peterson 2001, Thompson 2003, Pollock et al. 2004, Kissling and Garton 2006).

Using one observer for all sampling would not eliminate observer bias but it would make it more consistent. However, this is an impractical solution in most situations, especially for monitoring large areas over extended time periods. Extensive training in and the use of standardized sampling protocols is therefore the best way to manage observer bias and indeed it has been reported that for most common fish survey methods adequate training reduces observer bias significantly (Roper and Scarnecchia 1995, Bonar and Hubert 2002, Archer et al. 2004).

To control for bias associated with imperfect capture efficiency there are two approaches: literature review and independent quantification of capture efficiency of the target species. In some cases there has been considerable literature published on the capture efficiencies of common survey techniques on widely distributed species of management concern (Peterson and Cederholm 1984, Hillman et al. 1992, Rodgers et al. 1992). Appendix 2 provides a summary of the reported capture efficiencies of the sample methods recommended in this guide.

A more appropriate approach is to conduct a test of the capture efficiency of the specific survey method on the target population of interest, under the conditions likely to be present during the survey. A common technique used to assess capture efficiency is adding marked fish to a sample unit prior to sampling (Peterson et al. 2004, Rosenberger and Dunham 2005, Thurow et al. 2006). This is usually accomplished by first using block nets to close the sample unit and then using one-pass electrofishing to capture a group of fish. These fish are then marked and replaced into the sample unit (dispersing them evenly throughout the unit) and left to acclimatize for 12-24 hours. Then the sample method that will be used for the survey is used (this could be one-pass electrofishing or another method). The capture efficiency can then be calculated as the percentage of marked fish captured compared to all the marked fish released into the sample unit. To increase the accuracy of the capture efficiency estimate double block nets can be used to estimate the rate fish may be escaping the “closed” sample unit (Rosenberger and Dunham 2005).

We recommend that, regardless of the technique or survey design, minimum levels of training be provided for survey crews and that the training be coordinated at the regional level (see Section 2.4.2 for more detail). We also recommend that the estimated capture efficiencies reported in this guide be used when time and budget limit the ability of managers to independently estimate capture efficiencies for individual projects. If local knowledge indicates other capture efficiencies are more appropriate then they should override the ones recommended in this guide. If possible, tests of capture efficiency should be periodically conducted, especially for large projects with diverse habitat, or for projects involving threatened and endangered species where very precise estimates are required.

Estimating Sample Size

For any survey a decision will have to be made regarding the appropriate number of sample units to sample. This is generally referred to as the sample size (Elzinga et al. 1998, Thompson et al. 1998). Elzinga et al. (1998) provides a detailed description of how to determine sample sizes and we summarize the main points here. We also reproduce the formulas for determining sample size provided by Elzinga et al. (1998) in Appendix 3 and provide fisheries related examples.

Above all the sample size should be determined by the project objectives. The more confident you want to be about your parameter estimates, or the smaller the changes in population levels you want to be able to detect, the larger the sample size will need to be. You should also know something about the natural variability of the population you are sampling. Highly variable populations in space and time will require a larger sample size to produce equally precise estimates that more stable populations will provide. If the natural variation in a population (distribution and/or density) is unknown a pilot study is highly recommended (section 2.2.3).

In almost all cases involving fish surveys covered in this manual the sampling universe will be finite, that is there are a limited number of unique samples that can be collected. When this is the case, a correction factor known as the finite correction factor can be applied to sample size calculations which essentially reduces the number of samples required for a particular level of precision (Elzinga et al. 1998, Thompson et al. 1998, Krebs 1999). The finite correction factor should be applied if the number of sample units to be sampled is $\geq$ 5% of the total number within the sample frame. Table 3 summarizes the information required to calculate sample sizes for five typical fish survey objectives.

Table 3. Typical fish survey objectives and the information required to calculate sample sizes. Adapted from Elzinga et al. (1998). Detailed calculations and examples provided in Appendix 3.

Study Objectives	Statistics/Information Required	Definitions
1. Estimating mean number or total number of fish	- precision level (confidence interval width) - α (confidence level) - estimate of standard deviation	- e.g. mean ± 10% - how confident do you want to be that you are not making a Type I error of detecting a change when there was not one. Commonly set at α = 0.05 or 95% confidence level - how variable do you expect the samples to be
2. Detecting changes in a mean value between two time periods with temporary samples units	- power (1-ß) - α (confidence level) - size of change to detect - standard deviation	- how confident do you want to be that you are not making a Type II error of not detecting a change when there was one - same as above - e.g. detect a 10% change between sample periods - assumed to be equal between sample periods
3. Detecting changes in a mean value between two time periods with permanent or paired samples units	- α (confidence level) - power - size of change to detect - standard deviation	- same as above except standard deviation is calculated for differences between paired samples
4. Estimating a proportion	- precision level (confidence interval width) - α (confidence level) - estimate of the current frequency	- e.g. frequency of 25% ± 5% - same as above - if this is unknown use 0.5 (i.e. 50% frequency as conservative estimate)
5. Detecting a change between two time periods in a proportion using temporary sample units	- α (confidence level) - power - size of change to detect - estimate of the current frequency	- same as above
6. Detecting a change between two time periods in a proportion using permanent sample units	- α (confidence level) - power - size of change to detect - estimate of the change in frequency between two sample periods	- same as above except the expected change between two sample periods (transitions) has to be estimated initially (see Appendix 3 for details)

Temporal Aspects

The temporal aspects of the sample design will be determined by the specific objectives of the project, the target population, and the sample method selected. For example, night time snorkel surveys have been demonstrated to be more effective at sampling juvenile bull trout than day time surveys (Thurow 1996). However, night time surveys are more dangerous and may not be practical in remote areas.

For trend surveys to detect changes in distribution or abundance over time there are a variety of methods relating to the re-sampling cycle. Many long-term monitoring efforts are using some type of alternating panel design to increase the efficiency of large scale monitoring programs (Duncan and Kalton 1987, Lesser and Overton 1994, Thornton 1994, Manley et al. 2006). A biometrician should be consulted if a large scale long-term monitoring program is going to be established.

2.2.3 Pilot Studies and Prospective Power Analysis

Pilot studies can be very useful for estimating the relative amount of habitat types and critical population parameters such as mean threshold densities, variance, and detection probabilities. Pilot studies can also highlight access issues, other logistical constraints, potential sources of bias, and provide more accurate cost estimates of the proposed sampling scheme. The basic steps for conducting a pilot study are as follows:

1. Conduct a literature review of the target species. If information is very limited, interview other researchers with experience surveying the target species.
2. Delineate the likely sample frame and stratify the area if possible.
3. Conduct informal and purposive sampling using the best available techniques. Preferably use $\geq$ 2 techniques in a variety of habitats, seasons, and time of day to assess potential biases of different survey approaches.
4. Use the informal surveys to estimate habitat preferences, sample variance, and determine the most cost effective technique to conduct more statistically rigorous surveys.

The data gathered from a pilot study can also be used to estimate the statistical power of the proposed sample scheme to meet the objectives. Statistical power is a function of sample size, effect size, and significance level (Peterman 1990). See section 1.3.3 for more detail on statistical power.

<u>We recommend pilot studies be used at the beginning of all inventory and monitoring studies especially when there is limited information on a target population life history, distribution, threshold density, and/or capture susceptibility. We also recommend that statistical power be calculated a priori to make sure that the proposed sample sizes will meet the objectives (i.e. 80% probability of detecting presence above predetermined threshold density).</u>

2.3 Defining Inventory and Monitoring Objectives

Elzinga et al. (1998) outlines three general steps to defining inventory and monitoring objectives: determine the specific management objectives, outline the management response, and set the sampling objectives. Management objectives can be divided into two groups: target/threshold or change/trend (Elzinga et al. 1998). Target/threshold objectives set a level that will either be a desired state or red flag depending on whether current management actions are trying to reach or avoid the target/threshold. Change/trend objectives are designed to monitor change over a specified time. Change/trend objectives are useful when the rate of change a population may be experiencing is important to determine (i.e. how fast is a population increasing or decreasing). An appropriate management response(s) to the outcome of inventory and monitoring should be well defined prior to the start of a survey. If a management response is incorporated into the initial survey plan it is more likely to be supported and implemented (Elzinga et al. 1998). Management objectives should also be accompanied by sampling objectives. Sampling objectives need to clearly define the minimum levels of precision, Type I and II error rates, and magnitude of change the survey will be required to detect. See Sections 4.1, 5.1, and 6.1 for examples of unambiguous management objectives, management responses, and sample objectives for typical distribution, abundance, and trend surveys respectively.

2.3.1 Management Objectives

The broad management objective of fish inventory and monitoring surveys in wadeable streams is to determine the status or trend of populations on the National Forest System lands at the Forest scale. Specific objectives need to be clear, concise, and unambiguous (MacDonald et al. 1991). A management objective should have the following six components as described by (Elzinga et al. 1998, 2001):

- Species or indicator – what will be monitored.
- Location – where will the monitoring occur.
- Attribute – the specific aspect to be monitored (count of individuals, indices); should be sensitive to change, cost effective to monitor, and biologically meaningful.
- Action – the verb of the objective; should be to “maintain, limit, increase, or decrease”.
- Quantity/Status – is the measurable value the objective is trying to achieve (e.g. maintain 500 individuals, limit spread of rainbow trout to three watersheds, increase the population by 10%, or decrease the number of populations to six).
- Time Frame – the time required to meet the management objective; needs to be realistic and take in to account the biology of the target species; should try to keep time frames as short as possible because of the reality of funding sources and to promote regular assessment of the objectives.

2.3.2 Management Responses

Management responses are a statement of what management action will occur when a management objective is reached (or not reached) during the specified time frame. It is important to clearly define management responses prior to initiating the survey for the following reasons (Elzinga et al. 1998): if the management response is stated at the beginning of the project, managers, funding sources, and collaborators are more likely to support the implementation of the management response (i.e. no surprises) and if management options are limited, costly, or unknown this will be identified upfront and can be worked on concurrently with monitoring efforts.

2.3.3 Sampling Objectives

Sampling objectives set specific goals for the measurement of the particular value of interest (i.e. presence/absence, index value, population density). In essence they specify how accurately you wish to measure a value, and what level of risk you are willing to take that the estimate that you derive (i.e. population estimate) is not close to the true value (i.e. within 10% of the true value 95% of the time). We recommend specific levels of precision, power, trigger points, and scope of inference for distribution, abundance, and trend surveys in Sections 4.1.3, 5.1.3, and 6.1.3 respectively.

2.4 Data Collection

This section describes all the methods associated with randomly selecting sample units in the office and field, surveying the target population, recording and managing the data, and handling voucher specimens.

2.4.1 Data Collection Methods and Rationale

General Office Prep

An office information gathering exercise should be performed prior to each field survey. The office exercise should document previous sampling and species distributions, classify the valley and stream segment morphologies using map, aerial photography, and or GIS interpretation, and provide life history reviews for the target species. The precision of the existing data should be determined if possible, and where appropriate the existing data should be entered in the National Database.

Locating sample units

Locating sample units should be a two set process. First in the office the sample frame should be clearly marked on a map (preferably using GIS), and if necessary and possible, stratified with available map data. Then the total number of sample units per strata are sequentially numbered. The total number of sample units to be sampled is determined based on the degree of precision required and other factors including the presumed distribution and density of the target population. Sample units are then chosen using a random number table or computer program (Appendix 4). The geographic position of the selected sample units can then be uploaded to hand-held GPS units to facilitate their location in the field.

Layout and Marking

Most distribution surveys use a linear stream distance as a sample unit. The most common way to measure and mark stream distance is with a hip-chain or meter tape. A hip-chain can be used to measure the distance while sampling for fish. If a meter tapes is used, then the start and end points are usually measured and marked with flagging tape before sampling begins. Tape and hip-chain string should be removed from the area after sampling is completed.

Fish Survey Field Techniques

This manual provides specific protocols for conducting six common fish survey field methods which we grouped into A) intrusive and B) non-intrusive sampling methods: A) electrofishing, seining, plot/quadrat sampling and B) underwater observations (snorkel), redd counts, and minnow traps (Appendix 5). The protocols were adapted from pre-existing field methods primarily from Murphy and Willis (1996), Resource Inventory Committee (1997), American Fisheries Society (2007), and Johnson et al. (2007) fish survey manuals. Reviews of other state and provincial fish survey manuals were also conducted using the large number of protocols compiled by S. Bonar at the following site: http://www.ag.arizona.edu/srnr/research/coop/azfwru/scott/ under “links” and “Existing State/Provincial Standard Sampling Protocols for Freshwater Fishes”.

These methods can be applied for a wide range of objectives (distribution, abundance, and trend surveys).

Appendix 5 discusses the general safety considerations for working in the field. The remainder of Appendix 5 provides the following information for each sample method: advantages, disadvantages, appropriate situations for use, sampling operations, and fish handling procedures. Data recording procedures are discussed separately in section 2.4.4 and field data forms are provide in Appendix 6.

We also provide a dichotomous key that is adapted from Bonar et al. (1997) and Thompson et al. (1998) that will help survey coordinators choose the appropriate survey design and field techniques based on the objectives of the study and the conservation status of the fish species being surveyed (section 3).

Mark recapture, depletion, and distance sampling are specific types of techniques that are used for population estimation and are discussed in detail in section 5.3.1.

Voucher Specimens

Voucher specimens are important to collect, especially where field identification of a fish species is difficult, and when previously undocumented species are discovered in new areas. Proper preservation procedures in both the field and laboratory are important to maintain the quality of the collected specimens (RIC 1997). Proper permits should be obtained prior to sampling to ensure it is legal to collect particular fish species. If permitted, only one representative sample of rare or sensitive species should be collected. The following field and laboratory techniques for collecting voucher specimens are summarized from Barbour et al. (1999) and the Resource Inventory Committee (RIC 1997).

A 3.7% concentration of formaldehyde (known as formalin) is the most common substance used to fix fish tissues. To make a solution of buffered formalin, combine 1 part full strength formalin with 9 parts distilled water and add approximately 3 ml of borax (buffering agent) per liter of solution (McAllister, 1965). All fish must be killed prior to fixation. This is can be achieved by leaving the fish in high doses of the anaesthetizing solution until they stop breathing.

Each voucher specimen should be labeled with two labels. A waterproof label should be attached to the jaw or inserted into the mouth and a waterproof data label should be put on the outside of each jar. All labels must be written in pencil. The specimen label should contain the following information: fish identification number, species name of the fish, and the collection method. The data label should describe the stream name, alias, watershed/waterbody identifier (if applicable), reach number, site number, collection date, and the crew’s name(s) and a contact phone number (Appendix 6).

Pictures of all voucher specimens should be taken to document coloration and marking of the sample prior to fixation. All the voucher data should be entered into the national database and properly cross-referenced to the sample site for future reference.

Genetic Samples

Recent advances in molecular techniques have made it increasing more affordable to assess the genetic status of fish populations. Genetic assessments can be a powerful way to identify species, subspecies, and stocks that are morphologically very similar, and are also essential in assessing the impacts of hybridization between native and non-native species (Moran 2002, Nielsen and Sage 2002, Taylor et al. 2003). Non-lethal sampling of fish is now common place for obtaining tissues samples due to the advent of polymerase chain reaction (PCR) amplification which only requires very small samples of tissue (e.g. fin clips) (Moran et al. 1997, RIC 2001).

Genetic samples should be collected using the following procedures in areas where there are species that are hard to identify, there is uncertainty regarding the taxonomy of the species, or there is the potential for hybridization between species of management concern. Use surgical scissors to clip a small piece (1.0-0.5 mm^2) of a fin and place it in a minimum 1.5 mL of 95% ethanol. Micro-centrifuge tubes work well for collecting samples and should be filled with alcohol prior to starting sampling. The tube should be labeled with the fish number, site number, and date and a second label should be put inside the tube to ensure accurate identification. The samples can be stored at room temperature or kept in a freezer until DNA analysis is performed. Care should be taken to contact other researchers to make sure that the fin clips will not interfere with ongoing mark-recapture studies. Scales can also be used and they can be stored in typical scale envelops, but they may no provide the same amount of DNA.

Biological Study Ethics

Fish inventory and monitoring studies often involve capture and handling of individual fish. Regardless of the study objectives each researcher should attempt to limit the duress fish are subject to during capture and examination procedures. The least intrusive capture and examination techniques should be implemented whenever possible. The protocols in this manual are consistent with the guidelines set forth by ASIH et al. (1988), but each survey coordinator should review the guidelines before implementing a survey.

2.4.2 Personnel Qualifications and Training

Many large scale inventory and monitoring programs recognize the importance of qualifications and training (Roper and Scarnecchia 1995, RIC 1997, Bonar and Hubert 2002, Peterson et al. 2002, Archer et al. 2004). Review of field methods and results is essential to establishing and maintaining scientific credibility. Observers often have difficulty measuring certain attributes consistently, even when using the same protocol. These inconsistencies may be caused by insufficient training, (Wang et al. 1996, Whitacre 2004), inconsistent protocol application (Kondolf 1997, Whitacre 2004), or the use of protocols with imprecise measurement techniques, such as visual estimation (Kondolf and Li 1992, Ralph et al. 1994, Whitacre 2004). Training is thus extremely important for the data to have scientific credibility. We recommend that training be emphasized as an important and ongoing requirement of all surveys conducted on national forest lands and that a set of clear responsibilities be assigned to different members of the survey team.

Each survey should have a survey coordinator that is responsible for survey implementation, training, and performance evaluation. Each survey coordinator should have at least two 2 years of similar supervisory experience. If survey work is conducted under a contract, the survey coordinator will be responsible for contract inspection and monitoring of data quality. The survey coordinator should conduct, or organize annual training sessions prior to each field season to standardize understanding of survey objectives, protocols, fish identification, and data recording procedures. These training sessions should be regionally coordinated to reduce costs and to help maintain consistency in minimum crew standards across large areas. Training sessions should be conducted in streams where survey work will take place to familiarize crew members to the species and habitats they will likely encounter.

The field crews should consist of a minimum two-person team. No field crew members should work alone. Each crew should have one crew leader who has two years experience with fish surveys and a good knowledge of local fish species identification, habitat associations, and behavior.

2.4.3 Quality Control/Quality Assurance

Prior to any sampling the survey design should be reviewed by either an in-house statistician or another biologist within the Forest Service that is experienced in survey design. It would be preferable to have the reviewer be someone outside the district or regional office to help avoid local biases and to aid in coordination of sampling between different jurisdictions.

To ensure the quality of data collected, both the data itself and the techniques used to collect the data must be reviewed. For data to be considered complete, each data sheet must include the date, names of data collectors and their roles (measurer, data entry), and time of the collection. The information must be entered and checked before leaving the sampling site. A second independent crew should sample a random number of sites a second time to assess repeatability.

Data errors can generally fall into five types (Potyondy et al. In Press?): (1) errors of omission (e.g., not entering the date); (2) errors in data (e.g., field record measurement of 8 rather than 5); (3) errors of incorrect data entry (e.g., entering the incorrect unit of measure or data entry of a 7 rather than a 3); (4) errors of arithmetic (e.g., multiplying rather than dividing width by depth to get width/depth ratio); and (5) errors caused by sampling technique. Errors caused by sampling technique include equipment misuse (e.g., incorrectly calibrating data logger) or protocol misapplication (e.g., measuring along streambank instead of thalweg for reach length).

To produce of high quality data an emphasis must be placed on quality assurance procedures. Quality Assurance procedures can be administered to the inventory and monitoring process through both manual and automated techniques (Peterson and Wollrab 1999). In addition, field audits should be conducted to ensure that field data are collected and recorded to standards. Field audits are essential to ensure individual field crews are collecting data consistently and as intended by the standards. Verification of fish identification is also required. It is important for the survey coordinator to methodically check the quality of work during each of the phases of an inventory project. This practice will pick up errors as they are made, and prevent them from being carried through or compounded by successive steps in the inventory process. Following this review, the survey data can be entered into NRIS. The survey coordinator should then be responsible for arranging the statistical analyses of the data as needed.

2.4.4 Data Entry Forms

We have included a standardized field data form that should be used for all fish distribution surveys (Appendix 6). This standardized form will ensure that all the required information will be collected at each site and that the data will be compatible with the NIS database. Data input fields include Site ID number, visit date, geographic coordinates, location description, crew names, weather conditions, visit start time, and visit end time. For each site, a survey map will be created and maintained in a GIS. Using the most recent digital orthophotoquad as a base layer, the survey map should clearly display the sample site (reach) boundaries and access routes. A printed copy of this map (scale = 1:15,000: 8.5x11" sheet) will accompany the field data collection form during each visit. Use of this standardized map will facilitate orientation of surveyors within the reach and facilitate revisits to the site.

2.4.5 Logistics

There are numerous logistical considerations that are required to allow for an efficient and smooth implementation of a survey. Many of the following considerations will need to be addressed several months before any field surveys begin. The survey coordinator will develop an annual plan of operation that will address, at a minimum, the following logistical considerations for administering and conducting surveys (Vesely et al. 2006):

- Facility and equipment needs—acquisition and maintenance of field equipment.

- Transportation and access management—acquisition of vehicles, management of fueling and mechanical maintenance; arrangements for specialized licenses or authorization for use of all terrain vehicles; arrangements for spring clearing of road obstructions or chainsaw safety training for survey crews.
- Safety plan and equipment—development of a job hazard analysis for all aspects of surveys and review for needed revisions annually; acquisition of first-aid kits and training for survey crews.
- Radio communications—radio frequencies, procedures for contacting the dispatch center, radio communications procedures.
- Flagging and marking schemes—scheme for identifying transects and call stations; coordination with other resource units to avoid overlapping marking.
- Permits and handling procedures—contacting state and federal agencies to determine if special permits are required for the capture and handling of the target species (any threatened of endangered species as listed by the ESA will likely have permitting requirements).
- Agreements and memorandums of understanding (MOUs)—access agreements (if needed) with adjacent landowners; MOU with cooperating agencies and landowners.
- Contract administration (if work is done under contract)—frequency and mode of contact with contractor, inspection schedule, delivery schedule, and payment schedule.

2.5 Data Storage and Management

All data collected from fish distribution surveys will be stored in the Forest Service's NRIS. The details still need to be worked out for this ….

2.5.1 Data Cleaning Methods

All data collected during the survey (pre-field and field data) must be reviewed and checked for completeness and errors before entry into NRIS. Some data cleaning concerns that are specific to fish distribution surveys include accurate descriptions of the location of survey and proper species identification. GPS locations are mandatory for all sample locations; however, each sample location should also be mapped on 1: 24, 000 topographic maps and a site description should also be written for each site. The data is of little value if it can not be accurately attributed to a specific location of the ground. By recording multiple descriptions of the site location (gps, map, written) the location can be revisited by other people in the future with a high degree of confidence.

Proper species identification is also critical for distribution survey results to be meaningful. We therefore recommend that voucher specimens and photographs be collected whenever a survey is initiated in a new area or new species are discovered. Vouchers, photographs, and/or tissues samples are also useful where there is the potential for hybridization between species.

2.5.2 Database Structure

Needs to be developed still ….

Should describe the entire database including the variables collected in the field and any derived variables (and how they are calculated). This section should also describe the unit of measure and valid range of values for each variable.

2.5.3 Metadata Requirements

Metadata refers to “data about data” and standardized metadata is critical if survey data is to be transferable between current users and valuable to future investigations (Vesely et al. 2006). The metadata structure and content should be consistent with the Federal Geographic Data Committee (FGDC), Content Standard for Digital Geospatial Metadata (CSDGM), Biological Metadata Profile (FGDC and USGS 1999) and how these metadata standards are incorporated into NRIS. The CSDMG is a geospatial standard adopted by all federal agencies in 1995. It was created and is maintained by the FGDC. The Biological Metadata Profile was also developed by the FGDC in an effort to standardize the use of terms and definitions commonly used in the preparation of metadata for biological databases (Vesely et al. 2006).

2.6 Data Analysis

Once the data has been collected, reviewed for errors, and entered into the NRIS it is ready to be summarized and analyzed. Prior to any statistical analysis, the data should be explored using some simple data evaluation techniques such as tabulation and graphical displays. There are numerous references to aid people in developing graphical displays (Spear 1952, Tufte 1983, 1990); however, many are not readily available to forest managers. We recommend using Elzinga et al. (1998) as a reference for exploratory data analysis because of its comprehensive nature and availability as a free digital download (http://www.blm.gov/nstc/library/techref.htm). Survey coordinators should also review Section 3.4 in Vesely et al. (2006) for a good summary of how to approach data analysis of survey data. Another good source of data analysis approaches is RIC (1998) which is also available online (http://ilmbwww.gov.bc.ca/risc/pubs/tebiodiv/sif/index.htm). More specifics on data analysis are described under each specific survey objective.

2.7 Reporting

2.7.1 Expected Reports

Results from each survey will be summarized in an annual report using the standard format for scientific reports: Introduction, Methods, Results, and Discussion. The Introduction should present information and objectives specific to the forest region. The Methods section can briefly outline the methods described in this technical guide and Vesely et al. (2006), but it should also contain methods specific to the region: how the sampling frame was stratified, number of reaches in each stratum, and the range of dates for each survey visit. The Results and Discussion will be specific to the region. The annual report and subsequent publications are intended for use in the forest monitoring and evaluation reports of each forest in the region. The survey coordinator may also choose to publish results after one or more years of monitoring in a peer reviewed journal.

2.7.2 Reporting Schedule

The reporting schedule should follow the Forest Plan monitoring schedule of annual reports and five year summary reports.

Chapter 3.0 Selection an Appropriate Survey Design

3.1 Rational for Survey Design Selection

The following dichotomous key was adapted from keys developed by Bonar et al. (1997) and Thompson et al. (1998) to select appropriate surveys designs and sample methods for fish in wadeable streams. This key is based on two main factors: 1) the objectives of the study and 2) the conservation status of the fish species being surveyed. Survey design recommendations are detailed in Sections 2-5 of the guide. A description of the advantages, disadvantages, appropriate situations, and sampling procedures for common field sampling techniques are listed in Appendix #. This key can be used for any species of obligate freshwater fish species; however, we recognize that modifications of survey designs and sample techniques may be required as more field testing and research are conducted.

3.2 Dichotomous Key

****

If there is no reliable information regarding the life history characteristics or the factors that influence the distribution and abundance of the target species … STOP INVENTORY and MONITORING PLANNING and DEVELOP A PILOT STUDY (see section 2.2.3).

****

1a. Distribution or presence/absence information required (2)

1b. Abundance estimate required (5)

1c. Trend estimate required (13)

2a. Distribution or presence/absence does NOT have to be defined with specific degree of confidence (3)

2b. Distribution or Presence/absence DOES have to be defined with specific degree of confidence (4)

3a. Species is listed as threatened or endangered by the USFWS (LISTED) … conduct an INFORMAL SURVEY using the most appropriate NON-INTRUSIVE sample method listed in Appendix 5 (e.g. snorkel, redd counts). Presence of the target species satisfies the study objectives; however, absence of target species will have no measure of confidence. See Section 7.

3b. Species in NOT LISTED … conduct an INFORMAL SURVEY using the most appropriate sample method listed in Appendix 5. Presence of the target species satisfies the study objectives; however, absence of the target species will have no measure of confidence. See Section 7.

4a. The target species is LISTED … use the most appropriate NON-INTRUSIVE sample method listed in Appendix 5. Carefully define the sample frame and total number of sample units it contains. Estimate the sample size required to be 80% confident (power) of detecting target species at > 0.1 fish / 100 m sample unit (threshold density). See Section 4.2.1 and 4.2.2.

4b. The species is NOT LISTED … use the most appropriate sample method in Appendix 5 following the same survey design as 4a. See Section 4.2.1 and 4.2.2.

5a. Complete count of individuals required in all randomly chosen sample units … this will be difficult to accomplish in most wadeable streams. May be feasible with ichthyocides or other destructive methods. Consult regional managers about appropriate applications.

5b. Complete counts of all randomly chosen sample units not feasible or required … (6).

6a. A population estimate (absolute abundance) is NOT required … any relative abundance measure or index count can be used. However, we do NOT recommend relative abundance measures or index counts be used independently because there is no statistically valid way to relate the index to the true density without using a second technique.

6b. A population estimate (absolute abundance) is required (7)

7a. The target species is LISTED (8)

7b. The target species is NOT LISTED (10)

8a. Visibility is good and the target species and habitat conditions allow estimation of the perpendicular distance of each fish from a line transect or point … use DISTANCE method to determine population estimate (see Section 5.3.1 – Distance). Calculate the sample size sufficient to detect the abundance of a target population within 20% of the true number 80% of the time with $\alpha = 0.10$.

8b. Conditions not suitable for using DISTANCE method (9)

9a. Visibility is good … may be possible to use a combination of snorkel surveys and another non-intrusive method (e.g. minnow traps) to calibrate the snorkel counts. Electrofishing *may* be permitted for some listed species depending on population status, consult regional policies. See double sampling Section 5.3.2. Follow the same sample design as 8a.

9b. Visibility is poor … may be unable to conduct an unbiased population estimate. Consult regional managers for the appropriate methods.

10a. Capture efficiency is high (>0.3) (11)

10b. Capture efficiency is low (<0.3) and/or target population is at very low density (< 0.1 /sample unit) (12)

11a. At least 50 fish can be captured per sample unit, and there are sufficient funds to mark fish and revisit the site on more than one day ... use a MARK RECPATURE method with either batch marks (Peterson model – minimum two visits) or individual marks (CAPTURE model – minimum three visits). See Section 5.3.1. Calculate the sample size sufficient to detect the abundance of a target population within 20% of the true number 80% of the time with $\alpha = 0.10$.

11b. Funding limits sampling to one day per site maximum … use a DEPLETION method with a minimum of three passes. Use the same survey design as 11a.

12a. Population estimates are going to be very expensive for rare species especially with techniques that have low capture efficiencies. We recommend following the guidance of Thompson et al. (1998) in these situations and calculating the precision that can be achieved with the funds available. If the precision is unacceptably low the scope and objectives of the project should revised. It may also be possible to increase the acceptable Type I error rate (e.g. from 0.1 to 0.2) which can significantly reduce the sample size required and thereby reduce costs.

13a. The target species is LISTED (14)

13b. The target species is NOT LISTED (15)

14. Detection of trends in abundance will be exceeding difficult if intrusive sampling methods can not be applied. See 12a for recommendations for these situations.

15. We recommend using a MARK RECAPTURE method for trend monitoring as long as the assumptions of the model can be met or violations are minor as they will provide the best population estimates (lowest variance). DEPLETION and DOUBLE SAMPLING methods would also be appropriate. Calculate the sample size sufficient to detect a 20% change (one way if appropriate) in abundance of a target population 80% of the time with $\alpha = 0.10$. *** NOTE that detecting a trend may require a minimum of 5 years and possibly $\geq$ 10 years of monitoring.

Chapter 4.0 Strategies for Distribution Surveys

A central goal of ecology is to "… understand the patterns and causes of species distributions through time and space" (Angermeier et al. 2002). Distribution surveys are a common way to assess patterns and causes of species distribution and can result in models that can predict species occurrences in unsampled areas (Kruse et al. 1997, Rich et al. 2003, Oakes et al. 2005). The likelihood of detecting an individual species or detecting more species increases with the area sampled and the intensity of sampling within an area (MacArthur and Wilson 1967). Although distribution surveys are a powerful tool the results need to be interpreted cautiously. A major influence on the results of distribution surveys is the clumped distribution or "discontinuity" of stream fish. Angermeier et al. (2002) review the causes and consequences of discontinuity of survey results and highlight potential issues with interpretation of the results. For example, if the target species is at a low density its habitat preference may be difficult to discern because many preferred habitat units will not be occupied. Conversely, at very high densities the species preferred habitat may be saturated and individuals may be found in less preferred habitat. Survey coordinators should familiarize themselves with other potential factors that could bias distribution surveys.

The two types of distribution surveys described in this section are fish presence/absence surveys and range distribution surveys. Presence/absence surveys are typically used within a relatively confined area (e.g. valley segment to sub-basin scale), whereas, range distribution surveys are designed to more specifically determine the upstream and downstream distribution of a species of group of species over a larger area (multiple sub-basins or an entire watershed).

4.1 Objectives

The broad objective of a fish distribution survey in wadeable streams is to provide fish presence/absence and/or range distribution information necessary to evaluate fish populations at the National Forest scale. Presence/absence surveys can be used to simply identify whether a target species occurs within a predetermined area, or to describe habitat associations of the target species and co-occurrences with other species. Distribution surveys are primarily techniques that result in species lists and habitat associations that may then trigger management and/or regulatory actions (e.g. presence of salmonid fish can trigger specific riparian management actions, or the presence of an endangered species can trigger more detailed environmental assessments).

The following sections describe the general requirements for management objectives, management responses, and sampling objectives for distribution surveys. The following sections provide examples of how to write specific, unambiguous objectives and responses for distribution surveys likely to be performed on National Forests.

4.1.1 Management Objectives for Distribution Surveys

If little is known about a particular species on a National Forest, the most basic management objective for distribution surveys would be to determine if a species is present, or what the extent of its range is. If a reasonable amount of information already exists about a species (i.e. historical records of presence/absence and range), the next type of management objective for distribution surveys would be target/threshold objectives that describe a specific desired state that managers would like the population to reach within a set time frame (Elzinga et al. 1998).

4.1.2 Management Responses for Distribution Surveys

Management responses for distribution surveys entail plans to decrease, increase, or maintain the presence/absence of a species or its range. For example, suppose the survey results for a species of management concern demonstrate that it is not present within an area it was historically present in. The appropriate management response would be to implement management actions that would promote the re-establishment of the species into its historic range (e.g. barrier or invasive species removal, habitat restoration). Management responses should be clearly stated prior to monitoring and will be specific to each forest and region.

4.1.3 Sampling Objectives for Distribution Surveys

The following sections describe the recommended levels of precision and power, trigger point, and scope if inference that should be used when designing distribution surveys. We recognize that specific projects may have site specific objectives that will override there recommendations.

Precision and Power Levels for Distribution Surveys

A species absence from a sampling frame cannot be definitively proven because sampling methods are not 100% efficient in relatively large areas with complex habitat (Thompson et al. 1998, Bayley and Peterson 2001). Therefore, the objective of any distribution survey should be to describe the presence/absence of a species within a sample frame in terms of a statistical probability (Thompson et al. 1998). A threshold density needs to be specified prior to sampling in order to determine the probability a species is absent (see Section 1.3.3). The threshold density determines what density a population has to be below to be considered functionally absent. There is still considerable debate about what threshold density to use and how to set it for fisheries research (Green and Young 1993, Peterson et al. 2002, Hoffmann et al. 2005).

The desired level of precision will vary among projects, but we recommend that distribution surveys should have a minimum sample size sufficient to detect the presence of a target population within 20% of the true frequency (confidence interval) 80% of the time (power). We recommend that the interim threshold density should be set at 0.1 individuals per sample unit (100 m reach) until regional standards are developed (Hoffmann et al. 2005).

Trigger Point for Distribution Surveys

The setting of trigger points (where management action is initiated) will depend on the status of the target species, laws and regulations, and public involvement. Study designs should be developed to detect either an increase or decrease, but not both, as it is more efficient and cost effective (i.e. less sample sites required) for one-tailed (i.e. directional) survey designs.

<u>We recommend management actions should be considered when an estimated 20% change in frequency of occurrence or overall distribution is observed assuming the statistical power of the survey > 0.80 (Vesely et al. 2006).</u>

Scope of Inference for Distribution Surveys

The results of any survey can only be applied to areas beyond the specific sample points if a statistically rigorous sample design was used, which generally implies that each sample unit within the sample frame had a chance of being surveyed (Thompson et al. 1998). When planning distribution surveys biologists should try to coordinate their sampling with other agencies and adjacent forests to increase efficiency and their ability to apply the results to larger areas. Recognizing that inventories are likely an ongoing activity, it will be more efficient to plan sampling over the long-term instead of directing sampling at local areas to deal with issues of limited scope.

We recommend that randomized survey designs be employed whenever possible and activities be coordinated with other agencies and organizations to increase the efficiency and scope of distribution surveys.

4.1.4 Examples of Distribution Objectives

The following list provides examples of management objectives, management responses, and sampling objectives for the common types of distribution surveys likely to be conducted on National Forests.

Example 1

Management Objective – Determine if smelly darters are present within any part of the Slimy Creek watershed during the summer low flow period (July to September) in 2008.

Management Response – If smelly darters are present initiate a population estimate survey in the summer of 2009. If smelly darters are not detected in Slimy Creek, initiate a habitat survey or other environmental assessment to determine possible factors for its absence.

Sampling Objective – Obtain an estimate of presence of smelly darters within 20% of the true presence 80% of the time with a Type I error rate of 0.10. A threshold density of 0.1 smelly darters per sample unit will be used and absence will indicate that the darters are at a density $\leq$ 0.1 individuals per sample unit.

Example 2

Management Objective – Determine the range of adult (> 100 mm) smelly darters within Slimy Creek during the summer low flow period (July to September) in 2008.

Management Response – If the range of smelly darters has significantly decreased ($\alpha < 0.10$) from the historic range initiate a habitat survey to determine possible factors for its range contraction.

Sampling Objective - Obtain an estimate of range within 20% of the true range 80% of the time with a Type I error rate of 0.10.

4.2 Measures of Fish Distribution

Population measures that represent the survey objective must be selected to attain the objectives in a quantifiable way (Vesely et al. 2006). The following section describes some of the common population measures derived from distribution surveys. The section is divided into population measures derived from presence/absence and range/frequency of occurrence surveys.

4.2.1 Presence/Absence Measures

The minimum population measure required for distribution surveys is a list of species captured per sample frame. Presence can also be categorized by the life stage of the species which can indicate the life history activity associated with certain habitats (i.e. presence of fry may indicate spawning habitat). There is a scale of proof of presence for presence/absence surveys ranging from signs of presence (e.g. carcass, eggs, nest or redd), direct observation, capture of an individual, and finally to the collection of a voucher specimen. Categorizing the species detected as either native or non-native (i.e. introduced) has also become an increasingly important attribute to record because it can be used as a measure of the degree to which a fish community has been altered (Simon and Townsend 2003). Comparisons of the percentage of native fauna between areas could help managers focus restoration efforts for native species.

4.2.2 Species Range Occurrence Measures

For range and frequency of occurrence surveys more detail will need to be recorded. For range distribution surveys the most appropriate population measure is the linear stream distance occupied from the downstream minimum to the upstream maximum elevation of the target species presence. This measure will be important for determining if a target population range is stable or contracting/expanding, and in what direction. For frequency of occurrence surveys the population measure of interest is the percentage of sites occupied. The percentage of sites occupied could be further delineated by strata (i.e. stream type, habitat type, etc.) to give an indication of habitat preference (* Note this would require habitat types to be sampled in relation to their occurrence).

Species richness can also be calculated from distribution survey data. Species richness is simply the number of species present per sampling frame and can be compared to other sample frames as one measure of species diversity between areas (Vesely et al. 2006).

Habitat measures will be dependent on specific local objectives. Common habitat measures that are often used to stratify distribution surveys include gradient, temperature, and riffle/pool groups.

4.3 Field Methods for Estimating Fish Distribution

There are many methods that are appropriate for distribution surveys (Appendix 5). Passive or active techniques can be used for distribution surveys because in most cases the presence/absence of the target species is of primary concern (i.e. not abundance). However, when using passive methods (i.e. baited traps) care should be taken to make sure that fish are not being drawn into the sample frame thereby biasing the results.

<u>We recommend that either electrofishing, underwater (snorkel), and/or seine techniques be used for most distribution surveys. Appendix 5 outlines the steps necessary to conduct these and other surveys. There are a number of other well documented, though less commonly used, techniques that would be appropriate for distribution surveys where local conditions, or specific species, warrant their use.</u>

4.4 Data Analysis of Fish Distribution Surveys

This section will describe the recommended techniques for summarizing data collected during fish distribution surveys under subsections for each objective: presence/absence and range distribution.

4.4.1 Analysis, Synthesis, and Interpretation of Presence/Absence Data

Single Species

The data analysis, synthesis, and interpretation is relatively straight forward if the objective of the survey is to determine if a single species is present or absent in a particular watershed (e.g. sample frame). If the species was detected during a informal survey (e.g. visual observation from the bank or angling), then the detection and associated data (location, date, etc.) should be entered in the NRIS. The most useful way to present this type of data would be a map showing the location of the detection with other pertinent details as required by CSDGM.

The following data analysis, synthesis, and interpretation are recommended if a purposive survey did not detect the species in the area of interest, and a randomized design was developed to determine presence/absence:

1) *Prior* to sampling, calculate the number of sample units necessary to provide a 90% chance of detecting the species occurrence at an estimated minimum threshold density (see Appendix 3 for how to calculate the sample size).

2) If the species is detected before all the sample units are surveyed the survey coordinator can choose to stop sampling (to reduce costs) or complete the survey to allow for more data gathering.
3) Calculate the frequency of occurrence by dividing the total number of sample units where the species was present by the total number of sample units surveyed.
4) If the species was not detected, report the finding as "species X was not detected and there is a 90% chance that it is not present at densities > the minimum density threshold specified in the sample size estimate."

The underlying assumptions of these analyses are that the distribution of the species of interest approximates a negative binomial (i.e. clumped) or poisson (i.e. random) distribution. Data gathered during the distribution study should be used to determine which distribution best represents the data, especially if the distribution of the target population was not known prior to sampling (i.e. previous study or pilot study data tested). If neither the negative binomial or poisson distribution fits the data a statistician should be consulted to determine the appropriate models to use (also see Krebs 1999). The GLM MIX procedure in SAS is a powerful new tool that allows the user to specify a variety of data distributions for analyzing data.

Multiple Species

If presence/absence surveys are being conducted for multiple species then the most common methods for analysis are:

1. Making a species list of the species present in the sample and comparing that list to a list of the species that were expected to occur based on literature reviews and local knowledge. Discrepancies between the two lists can highlight deficiencies in the existing data, changes in historical occurrences, establishment of non-native species, and the presence of new species or range expansions.
2. Determining species richness (i.e. the total number of species per habitat or strata). Species richness can be compared between studies only if the same sample methods and same sample sizes are used. To compare species richness between studies of different sample sizes some form of standardization needs to be used. Krebs (1999) summarize several techniques that can be used to standardize the sample sizes. We provide a link to the program SPECIES DIVERSITY that Krebs (1999) recommends to perform one of these transformations (Appendix 7).
3. Species diversity can also be calculated from the distribution data of multiple species (if the total number of each species is also recorded). The two common measure of species diversity are the Shannon-Wiener and Simpson indices and Krebs (1999) provides a good list of recommendations for analyzing and summarizing species diversity data (see Krebs 1999 p. 451).

4.4.2 Analysis, Synthesis, and Interpretation of Range Distribution Data

The range distribution of a species or group of species can be measured by its size, shape, orientation, and internal structure (Brown et al. 1996). Vesely et al. (2006) recommends that before analyzing the geographic range of a species managers will need to decide whether they are interested in the full geographic range or only the specific areas where the species occurs. These two types of areas are defined as “extent of occurrence” and “area of occupancy” respectively (Gaston 1991). If the extent of occurrence is the objective of the range distribution survey, then simple maps showing the overall extent are the most common method of presenting the data. This type of information would be particularly important if a species was recently found in a new area. New range maps could indicate that the species may be much more wide spread than previously thought, and the new range maps could encourage further surveys in other areas or jurisdictions.

More often, the area of occupancy within in a species range will be the focus of range distribution surveys. The objective in this case is to display all known occurrences spatially and infer a geographic distribution. Comparisons of historic and present areas of occupancy can determine if they are contracting or expanding. Evaluations of the area of occupancy also give more detail than extent of occurrence, and can be used to reveal changes in the internal structure (i.e. size of holes and fragments) of a species range which can indicate the causes of range contraction or expansion. These data can also be used to calculate the detection probability and probability of occurrence for a species (Mackenzie et al. 2002, Stanley and Royle 2005, Kissling and Garton 2006). This information can be incorporated into mapping products to indicate the confidence managers have on survey results (i.e. streams can be color-coded based on the confidence level of the presence of a particular species).

Caution should be exercised when using historic range maps, especially when comparing them to current range maps. There may be differences between the two maps that are not ecological meaningful, but instead represent differences in the effort expended to create the maps (and the underlying survey data used to create the maps), or the mapping rules, scale, and resolution (Brown et al. 1996, Vesely et al. 2006).

4.5 Analysis Tools for Distribution Surveys

Appendix 7 provides a variety of statistical software packages that can be used to compare species richness between areas (when sample sizes are not the same) and calculate a variety of species diversity indices. The packages also contain information regarding the appropriateness of each tool for particular situations and the models underlying assumptions.

The analysis of presence/absence data can be greatly enhanced with the use of geographic information systems (GIS). It is beyond the scope of this manual to go into the detail of how to use GIS to present and analyze presence/absence data and we recommend a GIS technician be consulted prior to any data collection so that they are aware of the survey objectives and can make recommendations regarding possible analysis and presentation of the data. The following papers demonstrate some of the appropriate uses of GIS for analysis of presence/absence data: (Dunham et al. 1999, Vander Zanden et al. 2004, Fransen et al. 2006).

Chapter 5.0 Strategies for Abundance Surveys

This chapter outlines the objectives, field techniques, and data analysis required for abundance surveys. Abundance surveys can be either counts of individuals or relative counts of individuals or indices of individuals. Counts of individuals per unit area over some specified time period and are often referred to as absolute density (Krebs 1999). We will use the general phrase population abundance to refer to any measure of population size or density. There are three types of population abundance surveys: census, population estimate, and relative abundance (Thompson et al. 1998, Krebs 1999). A census is a complete count of all individuals in a defined area, during a defined period, and is not synonymous with a survey, which is an incomplete count of individuals (Thompson et al. 1998). A census is virtually impossible to conduct, even in the smallest streams, and is therefore rarely attempted. We will not review census methods for fish because they usually involve lethal sampling techniques, but see Boccardy and Copper (1963), Jacobs and Swink (1982), and Metzger and Shafland (1986) for examples.

Population estimates include any repeated count (minimum of two) of individuals per unit area where capture efficiency can be estimated (Elzinga et al. 1998, Thompson et al. 1998, Krebs 1999). Relative density measures determine the relative density of one population compared to another population(s). Population estimates, also called density estimates, are more expensive and time consuming than relative abundance estimates, and are typically used when more precise population estimates are required. For example, population estimate surveys are often used for harvested populations, endangered species recovery, or when relating abundance estimates to some type of vital statistic (i.e. reproductive rate) (Krebs 1999).

Relative abundance surveys are an index that is either explicitly or implicitly assumed to be correlated to true population size, although this relationship is not always rigorously tested (Anderson 2001). All counts of individuals without adjustments for detection rates are indices of relative abundance (Thompson et al. 1998). Relative density of fish can also be obtained by counts of objects such as spawning redds (Al-Chokhachy et al. 2005), as well as counts of carcasses (Ketcham et al. 2005b), various types of trap counts (Stolnack et al. 2005), and catch per unit effort surveys (CPUE) (Adams et al. 2004, Quist et al. 2006a). In general, relative abundance surveys are less expensive than population estimate surveys because of the reduced sampling effort. Relative abundance surveys can be used to determine if the target species occurs at higher relative densities in one area versus another, or to describe habitat preferences of the target species (i.e. more individuals found in strata 1 than strata 2). Relative abundance surveys are primarily techniques that result in data that can be used to infer the preference of one site or habitat over another (e.g. high densities of spawning adults or redds in association with certain habitat characteristics compared to areas with low densities).

However, relative abundance measures have been criticized as being invalid without empirical estimates of the detection probability of the target individuals (Anderson 2001, 2003), but see Engeman (2003) for a rebuttal. This is especially true when the index being used has a particularly low detection rate. It has been well demonstrated that the efficiency of backpack electrofishing and snorkel surveys in small streams often have detection rates < 50% during single pass surveys (Habera et al. 1992, Rodgers et al. 1992, Anderson 1995, Ensign et al. 2002, Peterson et al. 2004, Thurow et al. 2006). Detection rates of index counts can only be reliably determined by some type of empirically derived conversion factor. Conversion factors are developed by the use of more accurate (and likely more expensive) techniques, such as using a known number of marked individuals (Rosenberger and Dunham 2005), or double sampling and the use of ratio estimation (Eberhardt and Simmons 1987, Hankin and Reeves 1988), to determine the detection rate of the less expensive index method.

5.1 Objectives

The objectives of a fish abundance survey in wadeable streams is to provide population estimates or relative abundance information necessary to evaluate fish populations on National Forest System lands at the National Forest scale. Population estimates are the most reliable and scientifically defensible way to assess fish abundance and should be used instead of relative abundance estimates in most situations where relatively accurate abundance measures are required for management.

5.1.1 Management Objectives for Abundance Surveys

If little is known about a particular species on a National Forest, a pilot study should be conducted prior to attempting an abundance survey. If a reasonable amount of information already exists about a species (i.e. historical records of presence/absence and range), the management objectives most applicable for abundance surveys would be target/threshold objectives that describe a specific desired state that managers would like the target population to be at within a set time frame (Elzinga et al. 1998). See section 4.1.4 for specific examples of target/threshold objectives for abundance surveys.

5.1.2 Management Responses for Abundance Surveys

Management responses for abundance surveys entail plans to decrease, increase, or maintain the target population. For example, if the survey results for a species of management concern demonstrate it is below a predetermined target/threshold density (e.g. 100 fish/km of stream), the appropriate management response may be to implement management actions that would promote an increase in the abundance of the target species via habitat restoration measures or invasive species removal. Management responses should be clearly stated prior to monitoring.

5.1.3 Sampling Objectives for Abundance Surveys

The following sections describe the recommended levels of precision and power, trigger point, and scope of inference that should be used when designing abundance surveys. We recognize that specific projects may have site specific objectives that will override these recommendations.

Precision and Power Levels for Abundance Surveys

We recognize that some stream systems and species groups may be, for all intents and purposes, impossible to survey for reliable, unbiased fish abundance estimates based on available techniques and our current understanding of the stream dynamics. As an example, in sand bed dominated streams of the Coastal Plains, Adams et al. (2004) found numerous species that varied dramatically over time and space with no apparent habitat associations. In these systems precise abundance estimates for individual species are likely not feasible and other measures such as guild analysis may be more appropriate (S. Adams, Forest Service Southern Research Station, personal communications).

<u>For other less specious and dynamic streams the desired level of precision will vary among projects, but we recommend that abundance surveys should have a minimum sample size sufficient to detect the abundance of a target population within 20% of the true number 80% of the time with $\alpha = 0.10$. This level of precision is a compromise between the cost of sampling more sites to increase</u>

precision and a recognition that populations of fish have an inherent natural variability which can confound our ability to detect changes over time.

Trigger Point for Abundance Surveys

The setting of trigger points (where management action is initiated) will depend on the status of the target species, laws and regulations, and public involvement. Study designs should be developed to detect either an increase or decrease, but not both, as it is more efficient and cost effective (i.e. less sample sites required) for one-tailed (i.e. directional) survey designs.

We recommend as a general rule, and for consistency, that management actions should be considered when an estimated 20% change in the population is observed (Vesely et al. 2006).

Scope of Inference for Abundance Surveys

The results of any survey can only be applied to areas outside the specific sample points if some type of probabilistic sample design was used (Thompson et al. 1998). When planning abundance surveys biologists should try to coordinate their sampling with other agencies and adjacent forests to increase efficiency and their ability to apply the results to larger areas. Recognizing that inventories are likely an ongoing activity, it will be more efficient to plan sampling over the long-term instead of directing sampling at local areas to deal with issues of limited scope.

An example of how a survey design can be expanded to increase its' over all scope and expand the area of statistical inference would be the following:

A watershed with three levels of management, Bureau of Land Management (BLM) manages lower reach, Power Company manages mid reach and reservoir, USFS manages upper watershed. If a probabilistic survey design was implemented throughout the entire watershed to determine the abundance of species X stratifying the watershed by areas likely to contain the species, it would be more efficient than each agency conducting its own abundance estimate by reach.

<u>We recommend that randomized survey designs be employed whenever possible and activities be coordinated with other agencies and organizations to increase the efficiency and scope of abundance surveys.</u>

5.1.4 Examples of Abundance Objectives

The following list provides examples of management objectives, management responses, and sampling objectives for the common types of abundance surveys likely to be conducted on National Forests.

Example 1

Management Objective – estimate the population size of adult (> 100 mm) smelly darters within the Slimy Creek watershed during the summer low flow period (July to September) in 2008.

Management Response – If the population estimate of smelly darters is below 1000 adults initiate the restoration efforts outlined in the smelly darter management plan in the summer of 2009.

Sampling Objective – Obtain a population estimate within 20% of the true abundance 80% of the time with a Type I error rate $\alpha = 0.10$.

Example 2

Management Objective – Determine the relative abundance of adult (> 100 mm) smelly darters within Slimy Creek watershed during the summer low flow period (July to September) in 2008.

Management Response – If the relative abundance is below the relative abundance of Disturbed Creek watershed implement the smelly darter management plan restoration efforts in Slimy Creek.

Sampling Objective - Obtain an estimate of relative abundance within 20% of the true range 80% of the time with a Type I error rate $\alpha = 0.10$. Also conduct a validation test of the relative abundance count to ensure that the detection rate is constant throughout the study site.

5.2 Measures of Fish Abundance

Population measures that represent the survey objective must be selected to attain the objectives in a quantifiable way (Vesely et al. 2006). The best measure of population abundance would be a census (i.e. complete count without bias or error). However, this would usually be an inefficient use of an agencies' budget, time, and resources. A well planned, probabilistic survey should be able to provide a reasonably precise population estimate that can aid management decisions at a much lower cost.

The two options for estimating population parameters are population estimates and indices of relative abundance. The following sections describe common population measures within these two broad categories of abundance estimates.

5.2.1 Measures of Population Estimates

Measures of population estimates are derived from direct counts of individuals and are expressed in terms of the number of individuals within the sample frame (total abundance) or the number of individuals per sample unit or area (i.e. density). These measures should always have some level of precision reported with them. Variance, standard errors, and/or confidence intervals are all appropriate measures of precision.

5.2.2 Measures of Relative Abundance

Measures of relative abundance are derived from incomplete counts of individuals, or counts of objects thought to be directly correlated to the true population number such as redd counts, carcasses, trap counts, and CPUE. These types of counts provide measures that are often directly compared between strata or study sites without ever extrapolating the estimates to population estimates. These measures are also used to convert the indices to an population estimate based on some type of mathematical relationship between the indices and the true abundance. The problem with these techniques is that the relationship between the indices and the true population abundance is rarely tested even though it is likely to be affected by different habitats, observers, time periods, species, ages, and other factors (Anderson 2001).

Common population measures of indices include the total number of fish per pass (i.e. electrofishing pass, snorkel count, seine haul, etc) (Kruse et al. 1998), redds per length of stream our stream area (Dunham and Davis 2001, Al-Chokhachy et al. 2005), total number of smolt out-migrants per trap night (Flosi et al. 1998, Negus 2003), carcasses per day (i.e. area-under the curve analysis) (Taylor 1996, Zhou 2002), of some other measure of CPUE (i.e. number of fish captured or observed per time period) (Simonson and Lyons 1995).

5.3 Field Methods for Estimating Fish Abundance

Complete counts of fish populations, like other wildlife species, are rare because all individuals in a sample unit can rarely be observed or captured (Elzinga et al.

1999, Thompson et al. 1998). Field techniques for estimating fish abundance either attempt to adjust for incomplete counts to produce population estimates, or they settle for measuring relative abundance (i.e. index counts). Techniques for population estimates (i.e. mark–recapture, depletion, and distance sampling) adjust for incomplete capture efficiency and can provide relatively unbiased estimates of true population size (Elzinga et al. 1998, Thompson et al. 1998). Relative abundance techniques assume that there is a relationship (i.e. linear) between the relative measure (e.g. number of fish caught in a single pass electrofishing sample, or number of adult spawners) and the true population abundance. Index techniques use counts of objects that are also presumed to have some relationship to the true fish abundance. Index counts are less common in fisheries studies, but redd (nests), carcass, adult escapement counts are examples.

Relative abundance estimates are very popular and are used extensively because they are cheaper and less time consuming than techniques for population estimates. However, there is growing skepticism about the usefulness of relative abundance measures, especially when the relationship between the relative abundance estimate and the true population has not been evaluated (Thompson et al. 1998, Anderson 2001, 2003, Anderson et al. 2003). We agree that relative abundance measures are potentially fraught with bias.

Field methods for population estimates are generally more expensive and time consuming than relative abundance and index estimates. However, population estimates may be more cost effective and practical in the long run if relatively precise estimates or population numbers or density are required for a survey (Thompson et al. 1998). This is especially true because of recent advances in fish marking technologies, the relative ease with which fish can be captured and marked, and the ability of population estimates to provide unbiased or nearly unbiased estimates of population parameters (Thompson et al. 1998).

We ***strongly*** recommend that either direct population estimate techniques be used (i.e. mark recapture or distance sampling), or that relative abundance methods only be used in combination with double sampling techniques to provide estimates of the capture efficiency and sampling bias (Eberhardt and Simmons 1987, Hankin and Reeves 1988, Schwarz and Seber 1999, Bart and Earnst 2002). See section 4.3.2 for more information on double sampling techniques. We also recommend that block nets be used at the upstream and downstream ends of each sample unit when population estimates are required.

5.3.1 Field Techniques for Population Estimates

Mark-recapture, depletion, and distance estimates are the three most commonly used field methods for determining population estimates (Ensign et al. 1995, Elzinga et al. 1998, Thompson et al. 1998). The assumptions, common issues, detailed field procedures, and data collection steps for each of these four techniques are outlined below. All of these techniques have potential problems and none of them can provide completely unbiased estimates of population size or density. However, all of these techniques can adjust for the incomplete sampling efficiency inherent in any sampling technique and provide relatively unbiased and precise estimates of population abundance in situations where the assumptions can be met, or violations of the assumptions are minor.

Mark-Recapture

One of the most common and powerful techniques for population estimates in fisheries is the mark-recapture method (Thompson et al. 1998). A main advantage of mark-recapture methods over other population estimates techniques is that they provide additional population information such as movement rates and survivorship (Hilderbrand and Kershner 2000, Boss and Richardson 2002, Labonne and Gaudin 2005). The additional information mark-recapture methods provide can give insight into the mechanisms of population change (Elzinga et al. 1998, Thompson et al. 1998, Krebs 1999). However, these insights do come at a cost as mark-recapture techniques are generally more time consuming and expensive than other population estimate techniques (Elzinga et al. 2001). The precision of population estimates for mark-recapture models can also be poor, especially if few fish can be marked and recaptured. Krebs (1999) caution that at least 50 individuals need to be captured during each session, and at least seven marked individuals need to be recaptured to produce relatively precise population estimate.

All mark-recapture studies require at least two capture sessions and involve the initial capture and marking of individuals, the release of marked individuals back into the population, and then re-sampling of the population of marked and unmarked individuals. There are two types of mark-recapture models: closed and open. Open models are typically used for estimating survival rates are beyond the scope of this manual. See reviews in Elzinga et al. (2001), Krebs (1999), and Thompson et al. (1998) for a discussion of open mark-recapture models. The two basic closed mark-recapture models we will discuss are the Peterson (also called the Lincoln) method and group of models in the program CAPTURE (Otis et al. 1978, White et al. 1982, Thompson et al. 1998).

Peterson Model

The Peterson model for closed populations has the following assumptions:

- the population is closed and that there are no births, deaths, emigration, or immigration during the survey
- all fish have the same probability of capture
- marking fish does not affect their catchability or survival
- fish do not lose their marks
- all marks are observed upon recapture.

These assumptions are often violated in fisheries surveys but can be somewhat mitigated for. Blocks nets can be used to effectively close off the sample unit of interest thereby maintaining a more or less closed population (Peterson et al. 2005). Double block nets can also be used to estimate the number of fish escaping the “closed site” (Rosenberger and Dunham 2005). Mark loss and observation is usually not an issue because fin clips (e.g. caudal) can be used because of the short duration of most mark-recapture population estimates. And if marked fish are redistributed throughout the sample unit and allowed to recover for 24 hours the probability of capture should remain relatively constant (Mesa and Schreck 1989, Peterson et al. 2004).

The Peterson formula is as follows (Krebs 1999):

$$\frac{N = (M+1)(C+1)}{(R+1)} \qquad \text{(Equation 1)}$$

where, N = population estimate, M = number of fish marked in the first sample, C = total number of fish captured in the second session, R = number of fish in the second session that are marked.

In general, sampling should strive to have M approximately equal C (Krebs 1999). The Peterson model generally provides unbiased population estimates when M+C $\geq$ N and almost unbiased estimates when R > 7 (Krebs 1999). However, $\geq$ 50% of the population has to be captured to obtain relatively precise estimates and it is critical that the desired level of precision and sample size requirements be specified prior to sampling. In Appendix 3 we reproduce sample size charts for estimating sample sizes for small (< 100) and large (> 100) populations within $\pm$ 50%, $\pm$ 25%, and $\pm$ 10% respectively with α = 0.10 as per Robson and Regier (1964). These confidence widths represent targets that are appropriate for pilot studies ($\pm$ 50%), management studies ($\pm$ 25%), and research studies (± 10%). Appendix 9 provides methods for calculating the confidence intervals on Peterson estimates as per Krebs (1999).

CAPTURE Model

The CAPTURE model is more complex than the Peterson model and requires a computer program to analyze the data. The CAPTURE model has the same assumptions as the Peterson method plus it requires at least four capture sessions, unique marks for each capture session or individual marks for each fish, and relatively constant capture effort (Otis et al. 1978, White et al. 1982). However, unlike the Peterson model it does not require that the capture probability is equal among individuals (Otis et al. 1978, White et al. 1982, Thompson et al. 1998). Thompson et al. (1998) recommend this model over the Peterson model because the CAPTURE model can account for the observed lack of equal capture probability of individual fish and between capture sessions (Habera et al. 1992, Ensign et al. 2002, Rosenberger and Dunham 2005). However, some researchers have estimated fish populations with relatively high precision using the more simple Peterson model (Peterson and Cederholm 1984, Rodgers et al. 1992). Krebs (1999) and Thompson et al. (1998) provide extensive reviews of the application of CAPTURE models to population estimation. Appendix 7 provides links to software locations and more detailed descriptions on its use.

Depletion

Depletion estimates, also known as removal estimates, are another very common method for obtaining population estimates (Zippin 1958, Schwarz and Seber 1999, Bryant 2000, Wyatt 2002, Sweka et al. 2006). All removal estimates require at least two capture sessions and all captured fish are removed from the stream during each session. The basic theory is that population estimates can be derived based on the decreasing number of fish that are captured during each session. Captured fish are usually held in live wells during sampling. An advantage depletion estimates have over mark-recapture methods is that they can be less expensive and require less time because you do not have to mark fish and wait for marked fish to recover and redistribute within the sample unit. However, like mark-recapture estimates, the precision of population estimates for removal estimates can also be poor, especially if the capture efficiency of the sample method is < 0.3 (Thompson et al. 1998). The main assumptions of removal methods are that the capture probability between individual fish and capture sessions are equal, that the number of fish captured in each successive session is less than the previous session, and that the populations are closed (Elzinga et al. 2001, Thompson et al. 1998). In general, removal methods require that the number of individuals removed from the population is large relative to the true population. There are two general types of removal models: regression based and maximum likelihood based.

Regression Models

Regression approaches, such as the Leslie (1939) and Ricker (1975), rely on the proportional relationship between the CPUE and the existing population size. If the assumptions of the model are met, a regression plot of accumulated catch (x axis) and CPUE (y-axis) should produce a straight line (Krebs 1999). The x intercept of this line is effectively the population estimate (Figure 3). However, the assumptions of the regression model are often violated because capture probabilities are rarely constant between individuals. Depletion derived populations estimates are usually biased low because capture probability generally decreases with each pass (reviewed in Elzinga et al. 2001, Krebs 1999, Thompson et al. 1998). The data can be tested to see if they violated the assumption of constant capture probability by plotting the log of CPUE against the accumulated effort (i.e. time or area sampled) (Ricker 1975). However, regression depletion estimates should only be used for rough population estimates (i.e. pilot studies with ± 50% confidence intervals)

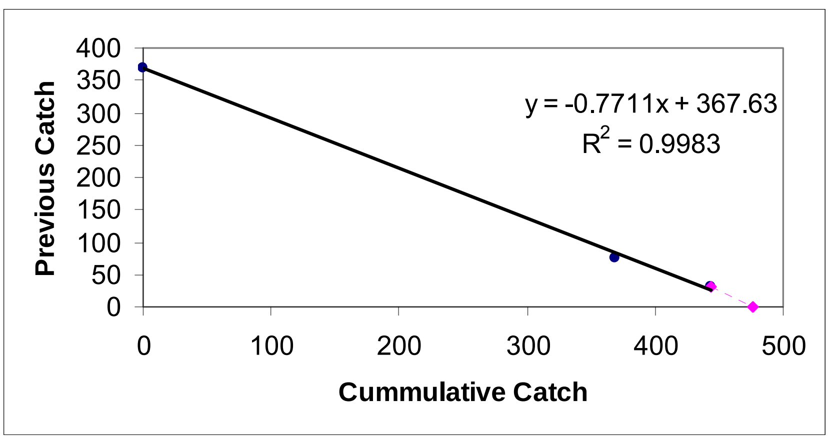


Figure 3. Example of a three-pass depletion estimate. The pink line depicts the predicted population size at the x intercept (477 fish) assuming a linear relationship.

Maximum Likelihood Models

Maximum likelihood models (MLM), also known as generalized removal models, were developed in response to the problems of satisfying the assumptions required for regression models (Krebs 1999). MLM still require closed populations and decreasing CPUE for each successive capture session, but they do not assume equal capture probability of individuals (Krebs 1999, Thompson et al. 1998). MLM models can still produce relatively poor estimates that are biased low especially when capture efficiency is low and only two removal sessions are used (Riley and Fausch 1992). Relatively precise population estimates have been reported if $\geq$ 3 removal sessions are used, with block nets, in relatively small streams where capture probability is high (Gowan and Fausch 1996). However, in a rigorous evaluation of multipass removal estimates, Peterson et al. (2004) showed that these techniques on average overestimated capture efficiency by almost 40% and underestimated population size by almost 90%. Peterson et al. (2004) attributed these poor results to influences of stream characteristics (area and complexity), fish species, and fish size. These factors were negatively related to the first pass capture efficiency and to the magnitude of the decrease in efficiency with each successive pass. Peterson et al. (2004) suggest that removal estimates be considered biased indices unless capture efficiency is explicitly determined with validation sampling and used to adjust the removal. VanDeventer and Platts (1989) provide the software package MICROFISH that can provide maximum likelihood populations estimates using depletion data (Appendix 7).

Distance

Distance sampling procedures have been used for many years in wildlife management but are relatively rare in fisheries studies (Krebs 1999, Thompson et al. 1998). Distance methods resolve around the surveyor being able to estimate the distance of animals from a point or line transect. These methods can provide good unbiased estimates because they use the visibility bias (animals further from the point or transect are more difficult to detect) of undetected animals is adjusted for by using the recorded distances between the observer and the detected animals (Elzinga et al. 2001). The assumptions of most distance methods are that animals can detected at a fixed position before they more (fright response), any animals at the point or on the line transect are detected, and distances are measured without error (Elzinga et al. 2001). Distances can be recorded into to categories (i.e. <5 m, 5-10 m, etc.) but at least five categories are required if exact distances are not used. At least 60 animals should be sighted to get a population estimate, which could severely limit the technique for rare species. The program DISTANCE can be used to analyze the data (Appendix 7).

Similarly to the quadrat count methods, distance sampling are likely only to be used in specialized cases where other techniques fail to provide precise population estimates. Small benthic fishes that use clear streams and are highly adapted to high velocity riffle habitat may be a situation where distance sampling can provide more accurate population estimates (Ensign et al. 1995).

5.3.2 Field Techniques for Estimating Relative Abundance

As discussed previously, population estimates are relatively time consuming and expensive. Therefore, a multitude of measures of relative abundance have been developed that are faster and cheaper to implement. Of course, faster and cheaper can often lead to estimates that are of little value for making management decisions (Anderson 2001). Some of the most common relative abundance measures used in fisheries management are single pass electrofishing, underwater observation, above water observation, minnow traps, angling, redd or nest counts, carcass counts, and plot/quadrat counts. Because these techniques are so widely used and can be applied to a variety of different objectives (i.e. can be used for distribution, abundance, and trend surveys) we describe the recommended field procedures for each of these techniques in Appendix 5.

<u>We recommend that the capture efficiency of any relative abundance measure be determined in one of the following ways: a literature review or double sampling. If the capture efficiency of the indices has been thoroughly evaluated (Bayley et al. 1989, Peterson and Rabeni 2001) for the habitat and species you are working on then the indices may be calibrated with the existing data. We caution however, that capture efficiency will likely vary spatially and temporally</u>

and in unpredictable ways, and it is advisable to use double sampling techniques to regularly assess the capture efficiency of any indices that is used.

Double sampling techniques require the use of a more reliable technique to determine the true abundance for comparison with the index count. One double sampling technique that has been used recently is adding a known number of marked fish to a sample unit (Rosenberger and Dunham 2005). Then when a index count is applied to sample unit (e.g. diver count), the proportion of marked fish detected will provide an estimate of capture efficiency. Double sampling only has to be conducted on a small portion of the proposed sample frame, and should be divided evenly among the different strata. See Bart and Earnst (2002), Eberhardt and Simmons (1987), Kissling and Garton (2006), and Schwartz and Seber (1999) Hankin and Reeves (1988) for a further review of double sampling strategies.

5.4 Data Analysis for Abundance Estimates

This section describes the recommended techniques for summarizing data collected during fish abundance surveys under subsections for each objective: population estimates and relative abundance.

5.4.1 Analysis, Synthesis, and Interpretation of Population Estimates

As described by Elzinga et al. (2001), there are only two instances when statistical analysis of count data are not required, or are inappropriate: for complete censuses and when the data are not derived from some form of randomized sample design. Assuming no measurement errors, census data can be presented as the true population value. Interpretation of census data will require an understanding of the biological significance of the result (i.e. does the census data indicate a healthy population or one under stress).

Purposive sampling (i.e. counts based on representative or convenience sampling) can not be validly assessed with statistical tests and the results are technically only applicable to the areas that were sampled because other sample units did not have a chance of being sampled (Thompson et al. 1998, Anderson et al. 2001, Anderson et al. 2003).

<u>It is for these reasons that we recommend using some type of randomized sample design whenever possible. For all partial counts with data derived from randomly designed surveys we recommend the following approach to data analysis, synthesis, and interpretation as described in more detail in Anderson et al. (2001), Elzinga et al. (1998, 2001), Krebs (1999), and Thompson et al. (1998).</u>

Exploratory Analysis

Abundance data are either count data or density data (also called measurement data) and as such the data are discrete, meaning they are non-continuous (i.e. you can't have a half a fish). Fish populations tend to be clumped and the among sample unit variation is usually high (reviewed in Thompson et al. 1998). Fish populations are typically clumped because of either habitat heterogeneity or their behavior (i.e. schooling). Therefore, the variance associated with fish counts tends to be larger than the mean (reviewed in Elzinga et al. 2001 and Thompson et al. 1998).

There are a variety of graphical means to reveal patterns in the data and for suggesting if the data are normally distributed, or for highlighting outliers (Elzinga et al. 1998). Plotting the individual data points (each fish count per sample unit for example) can also depict the distribution of the data better than just relying on the mean and the standard deviation values (Figure 4). Normally distributed data are a prerequisite to most parametric statistical tests (see below) and outliers can indicate possible errors in data entry or collection. Normal probability plots can be used to determine if the data approximate the normal distribution. If the data are not normally distributed you can use various transformations to make the data normal (log, arcsine, etc.), use non-parametric tests, or use parametric tests and ignore the non-normal distribution (some parametric tests are relatively robust to violation of non-normality). See the following section *statistical analysis* for further details.

Box plots and density plots are also useful graphical means of exploring the data. Figure 5-7 provide examples of normal probability plots and different kinds of box plots and density plots based on some hypothetical fish survey results for two different sites (Table 4). Detailed explanations of these graphical methods are given in Elzinga et al. (1998).

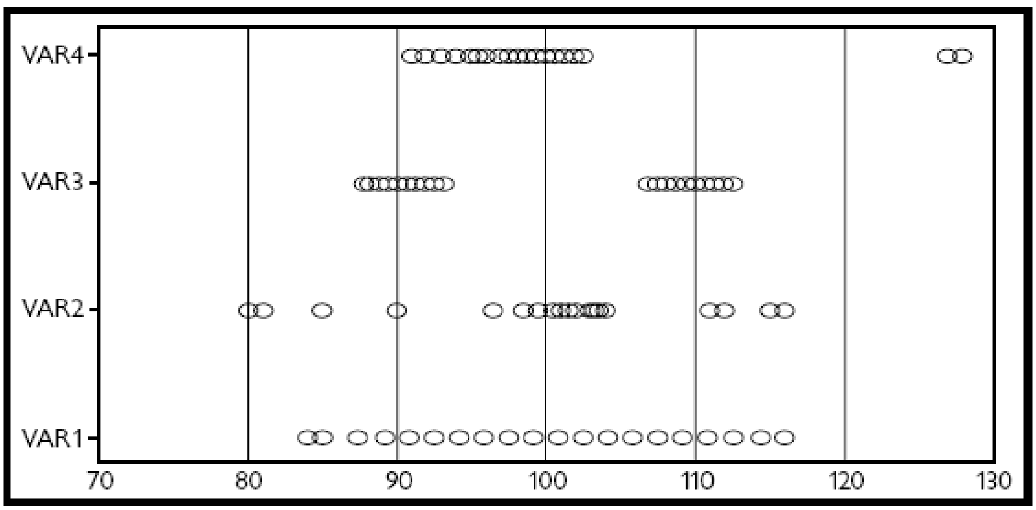


Figure 4. Four separate samples (n = 20), each with a mean of 100 and a standard deviation of 10. It is obvious from this figure that the mean and standard deviation alone can not reveal the distribution of the data. Figure from Elzinga et al. (1998).

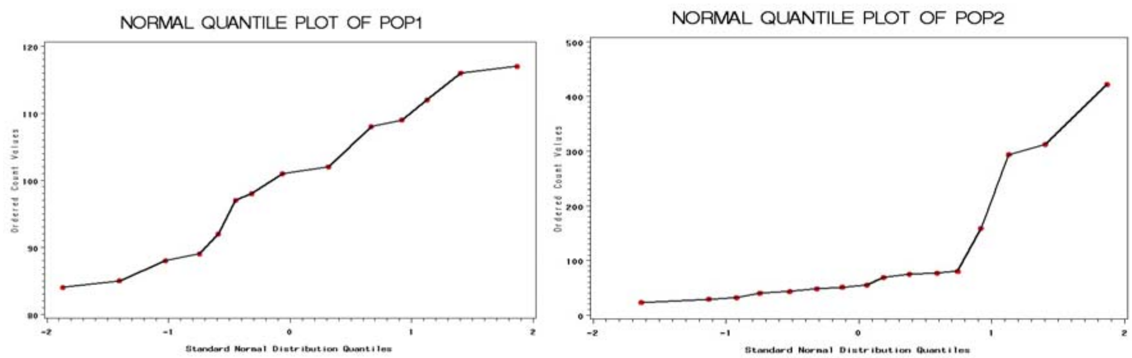
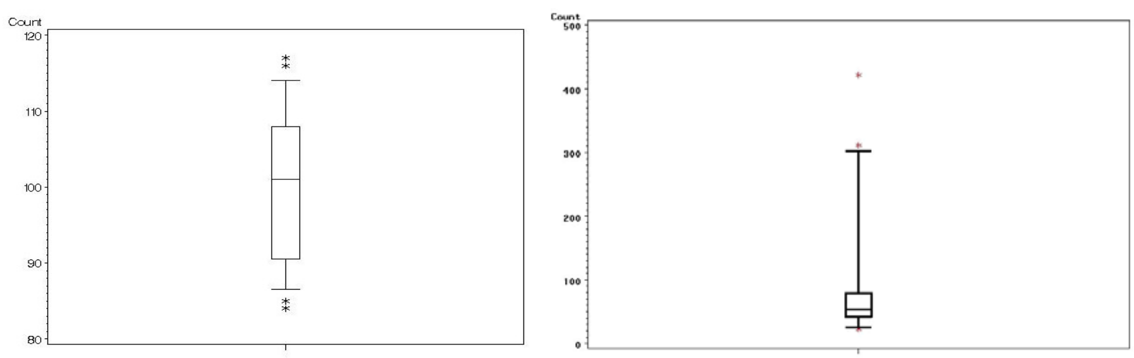
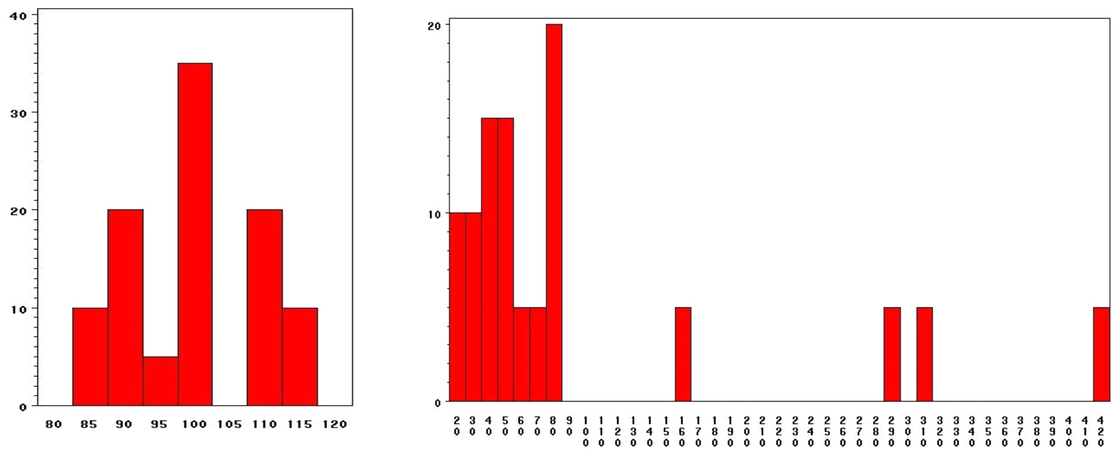


Figure 5. Example of normal probability plots for two hypothetical fish survey results (Table 4). Normally distributed data will form a relatively straight line (i.e. population 1) from the bottom left to the top right corner, whereas non-normal data will not form a straight line (i.e. population 2).



Population 1 Population 2

Figure 6. Example of box plots for two hypothetical fish survey results (Table 4). Line across box = median, top of the box = 75% percentile, bottom of the box = 25% percentile,



Population 1 Population 2

Figure 7. Example of density plots (frequency histograms) for two hypothetical fish survey results (Table 4).

Table 4. Data for samples from two hypothetical populations (n = 20). Population 1 has a relatively normal distribution whereas the data from population 2 is not normally distributed (see Figures 5-7).

Reach	Population 1	Population 2
1	84	23
2	101	23
3	108	29
4	92	32
5	117	40
6	102	43
7	97	43
8	88	48
9	108	51
10	112	51
11	102	55
12	101	69
13	116	75
14	89	75
15	101	77
16	98	80
17	109	159
18	102	293
19	85	312
20	88	422

Statistical Analysis

There are two basic types of analysis for count data: parameter estimation and significance tests (Elzinga et al. 1998). Parameter estimates are used to estimate a single independent sample (e.g. the number of fish/m^2 in stream X or the number of fish/m^2 in watershed Y). Significance tests are used to detect the difference between two or more counts or a change from one period to another.

Single Parameter Estimates

For single parameter estimates (i.e. means, total, proportions) the appropriate analysis is to estimate the precision of the estimate with confidence intervals (Elzinga et al. 1998). A common situation in fisheries where this would be appropriate is the following example:

The target/threshold management objective is to have at least 10,000 adult ($\geq$ 25 cm length) small mouth bass within the Coolwater Watershed basin each year from 2007-2012 throughout the current 5 year Forest Service Management Plan for the Green Acres Forest. The sampling objective is to be 95% confident that the estimate is within $\pm$ 500 fish of the target/threshold population. In the summer of 2007 a stratified random survey was conducted to estimate the population size of the entire watershed and the total number of smallmouth bass was estimated at 9,800. This total is only an estimate of the true number of fish present which was estimated by multiplying the mean number of bass per sample unit by the total number of sample units in the watershed. The precision of this estimate should be determined by calculating confidence intervals. See Appendix 9 for calculating confidence intervals.

Elzinga et al. (2001) have an excellent discussion of the interpretation of confidence intervals. They outline the four possible outcomes of calculating confidence intervals: 1) the threshold level is not crossed by either the parameter estimate or the confidence interval (there are less fish than your threshold), 2) the threshold has been crossed by both the parameter estimate and the confidence intervals (there are more fish than the threshold), 3) both the parameter estimate and the upper bound of the confidence interval crossed the threshold, but the lower bound of the confidence interval did not (may have exceeded the threshold but not as confident as situation 2), and 4) the parameter estimate does not exceed the threshold but the upper bound of the confidence interval does (may have exceeded the threshold value but even less likely than situation 3).

If in the above example we calculated a confidence interval for the total number of bass as 9200 ± 1000 bass this would be similar to situation 4 above. This means that there is a 95% chance that the total number of bass in the Coolwater Watershed is somewhere between 8200 and 10200 bass. There is also a 5% chance that there are less than 8200 or more than 10200 bass. In this scenario the upper bound of the confidence interval has crossed the threshold value set as a management objective, but only by 400 fish whereas the lower bound of the confidence interval is 1,600 fish below the threshold value. This scenario suggests that is more likely that the threshold value has not been met.

To avoid situations like 3 and 4 above, well planned surveys are required that have sufficient sample sizes, minimal variance in the sample data, and appropriate confidence levels (i.e. if a species is highly variable and difficult to census having 95% confidence levels may be unrealistic – perhaps 80% would be more suitable).

The examples above assume that a population estimate has already been computed. We will not review here the analysis techniques for population estimates since there are a variety of field methods that could be used for population estimates (i.e. mark-recapture, depletion, quadrat, and distance sampling) and further assumptions that have to be made when using a particular method (i.e. open or closed models). Section 4.5 describes the analysis tools that can be used for population estimates depending on the field technique used and the assumptions of the survey design.

Significance Tests

If the management objective is to determine if two or more estimates of density or total number of fish differ, then the appropriate statistical analysis requires significance testing or hypothesis testing (Elzinga et al. 1998). If the management objective is to determine changes in abundance within sample areas or sites over time, it requires trend analysis that is covered in Chapter 5.

Anderson et al. (2001), Elzinga et al. (1999, 2001), Krebs (1999), Romesburg (1981), Thompson et al. (1998) and many others caution that prior to any significance testing an a priori null hypothesis should be clearly stated. For example, if monitoring is initiated to determine if forest harvesting along side stream X is decreasing fish numbers per km compared the numbers of fish per km in stream Y that has no stream side harvesting, an appropriate null hypothesis would be “the number of fish per km in stream X and Y are the same.” Of course this assumes that we had some biological reason to believe that the two streams should have similar numbers of fish per km to begin with.

Analysis of the data depends on the distribution of the data (i.e. normally distributed) and the type of data collected (count, frequency, etc.). Elzinga et al. (1998) describe the appropriate statistical tests for abundance (count) data using parametric and non-parametric tests (Table 5). The decision to choose parametric tests depends of meeting the following assumptions:

- Population being sampled has a normal distribution,
- The variances of sample populations are the same (homogeneity), and
- Sample units are drawn randomly.

Table 5. Specific statistical significance tests and the types of parametric and non-parametric tests that should be used. Adapted from Elzinga et al. (1998) Table 11.2, p. 256.

Purpose of the test	**Temporary/ Permanent Samples***	**Type of Data**	**Parametric test**	**Non-parametric test**
Change between two years	Temporary	Not frequency	Independent-sample t-test	Mann-Whitney U test
Change between two years	Permanent	Not frequency	Paired t-test	Wilcoxin's signed rank test
Change between two years	Temporary	Frequency		Chi-Square test (2x2 contingency table)
Change between two years	Permanent	Frequency		McNemar's test
Change between three or more years	Temporary	Not frequency	Analysis of Variance (ANOVA)	Kruskal-Wallis test; Mann-Whitney U test
Change between three or more years	Permanent	Not frequency	Repeated Measures Analysis of Variance	Friedman's test; Wilcoxin's test
Change between three or more years	Temporary	Frequency		Chi-square test (2x 3 contingency table)

However, Elzinga et al. (1998) review the consequences of violating these assumptions and conclude that T-tests and analysis of variance (ANOVA) are relatively robust to modest violations. If the violations of the assumptions are severe use of non-parametric alternatives are likely the best alternative.

Significance tests provide a test statistic and a p-value. For example, the T-test computes a t statistic and an ANVOA computes an F statistic. The test statistic is a measure of the difference between the two samples (e.g. the total number of fish in stream X compared to stream Y). The p-value is the probability of obtaining a particular test statistic as large, or larger than the one computed when there is no difference between the two populations (Freedman et al. 1998). To determine if there is a statistical difference between two populations a threshold p value should be specified before the samples are collected. A traditional p value threshold is often 0.05. Therefore, if the p value associated with a test statistic is 0.03, we can conclude that the two populations are different and there is a 3% chance that we are wrong (i.e. we have committed a Type I error and assumed there was a difference between the populations when there was not).

Caution should be used when interpreting significance test results because statistical significance does not always equal biological significance (Krebs 1999). Statistical significance is determined by sample size, difference in populations, and efficiency of the sample design and therefore, large sample sizes alone can produce statistical differences that have little biological meaning. Also, it is important to note that the magnitude of the p value should not be interpreted as a measure of the actual effect size (Anderson et al. 2001). In the example above it is possible to get a significant p value when comparing the abundance of fish in stream X and stream Y based purely on large sample sizes. Therefore, without some measure of the true magnitude of the difference (i.e. the mean difference in the number of fish per km and a standard error estimate). As Anderson et al. (2001) suggest, naked or nearly naked p values are of little use in interpreting the results of a significance test.

Another common problem in monitoring studies is not finding a statistical or biological change in monitoring studies due to poorly designed surveys with low power (Peterman and Bradford 1987 , Maxwell and Jennings 2005). The power of a survey should be determined a prior because post-hoc power analysis are not statistically valid (Gerard et al. 1998, Anderson et al. 2001, Hoenig and Heisey 2001). See Appendix 7 for links to computer programs and manuals for methods and descriptions of how to calculate the power of a survey design.

Tests of statistical differences between populations need to be conducted differently depending on whether the samples are independent or paired. Independent samples are chosen randomly during each survey. For example, when monitoring stream X to determine the total number of fish each year, new random sample sites chosen each year would be considered independent. If however, the sites were chosen randomly during the first year of the study, and then re-visited each subsequent year, the study would be considered a paired study. Table 5 describes the proper statistical tests to use depending on the purpose of the statistical test and the survey design.

Two other considerations that are required when conducting significance tests are whether the tests should be one or two-tailed and whether the finite correction factor should be used (Elzinga et al. 1998; Krebs 1999). A one-tailed statistical test is appropriate if alternate hypothesis is directional. For example, we may only be concerned if population X is smaller than population Y. Therefore, when setting up the survey the null hypothesis would be that population X and Y are the same, and the alternate hypothesis would be that population X is smaller than population Y. If however, we wanted to know if population X was smaller or larger than population Y, then a two-tailed test would be appropriate. The null hypothesis would still be the same, but the alternate hypothesis would be that population X is either smaller or larger than population Y.

The finite population correction factor (FPC) is applied to the results of the significance test whenever > 5% of the entire population has been sampled (Zar 1984). For example, if a population estimate on stream X that was 100 km long was calculated based on a sample of 10 km (i.e. 10% of the total length) then the FPC should be applied. The finite correction factor rewards large sample sizes by increasing the test statistic and thereby increasing the ability of the test to detect differences between populations (Elzinga et al. 1998; Krebs 1999).

5.4.2 Analysis, Synthesis, and Interpretation for Relative Abundance Estimates

The same analysis and synthesis of relative abundance data can be conducted as for population estimate data. Confidence intervals can be constructed around single pass electrofishing results, redd counts, and other relative abundance measures and significance tests can be performed on measures from different strata or study streams. However, the interpretation on relative abundance measures (indices) can be more complicated or even worse, misleading (Anderson 2001, 2003). As discussed previously, without some type of validation of index counts, there is no way of knowing if your results are a true reflection of the actual population number (Eberhardt and Simmons 1987, Schwarz and Seber 1999, Bart and Earnst 2002, Tracey et al. 2005, Marchandeau et al. 2006, Toms et al. 2006).

Likely one of the best ways to calibrate an index count based on fish counts (i.e. single pass electrofishing, snorkel surveys, seine surveys, etc.) is to close of the sample unit with block nets, capture, mark, and release an initial group of fish, and then use the index technique as planned (i.e. single pass electrofishing). In this type of scenario there is now a known number of fish in the sample unit and the number of marked fish captured or observed with the index technique can be used to estimate the capture efficiency of the index technique and more importantly, determine if the index technique has an consistent capture efficiency for different species, life stages, and habitat types. Further descriptions of these techniques are described in Peterson et al. (2004) and Rosenberger and Dunham (2005).

5.5 Analysis Tools for Abundance Estimates

There are a wide variety of statistical tools available to analyze abundance data from common and relatively simple programs like Excel and SAS JMP, to more complex and powerful statistical software and share ware such as SAS, SYSTAT, SPSS, and wide variety of free statistical packages online (i.e. R).

Some tools, such as SAS have new powerful statistical tools such as GLM MIX that allow the user to specify the distribution of the data and avoid many of the assumptions that are required with other tools (i.e. normally distributed data).

Chapter 6.0 Strategies for Trend Surveys

This chapter outlines the objectives, field techniques, and data analysis required for monitoring trends in fish distribution and abundance. Thompson et al. (1998) define a goal of population monitoring " … to detect an important change, in both the magnitude and direction, in the average number of animals over a defined time period." Often the goal of trend analysis is to document a change in population abundance due to restoration activities (Raborn and Schramm 2003, Shields et al. 2003) or perceived negative impacts from development activities such as forest harvesting (Bjornn and Reiser 1991, Loftus and Flather 2000). Any of the population measures discussed so far in this manual (e.g. distribution, and population estimate, relative abundance, and indices) can be used to monitor changes in population status over time, but there are strengths and weaknesses associated with each approach (Vesely et al. 2006).

Trend can be measured over a variety of time periods (e.g. seasonally, annually, etc.) and spatial scales (e.g. site specific to regional) (Urquhart et al. 1998). This manual focuses on detecting trend at the Forest scale which equates to areas of approximately 400,000–500,000 hectares. We will focus on the strategies required to assess trend at this scale on an annual basis (year to year changes); however, the concepts discussed here are generally applicable to other spatial and temporal scales.

We briefly review several important concepts and issues that relate to trend survey design and analysis below, and recommend that survey coordinators review Adams et al. (2004), Larsen et al. (2001), Roper et al. (2003), Thomas (1996), Thompson et al. (1998) Chapter 5, and Urquhart et al. (1998) for more details on the challenges of trend analysis.

Types of Trends

If you plot year on the x-axis and the abundance of fish measured annually at a site or series of sites (average count) on the y-axis you can expect to see one of four common types of trends data: random, linear (upward or downward), exponential (upward or downward), and cyclic (Figure 8). Variance from a variety of sources, collectively called “noise”, can mask true trends in the data (see below for the sources of variance). The more variance there is associated with the trend data the more difficult and/or costly it will be to detect changes in the population abundance.

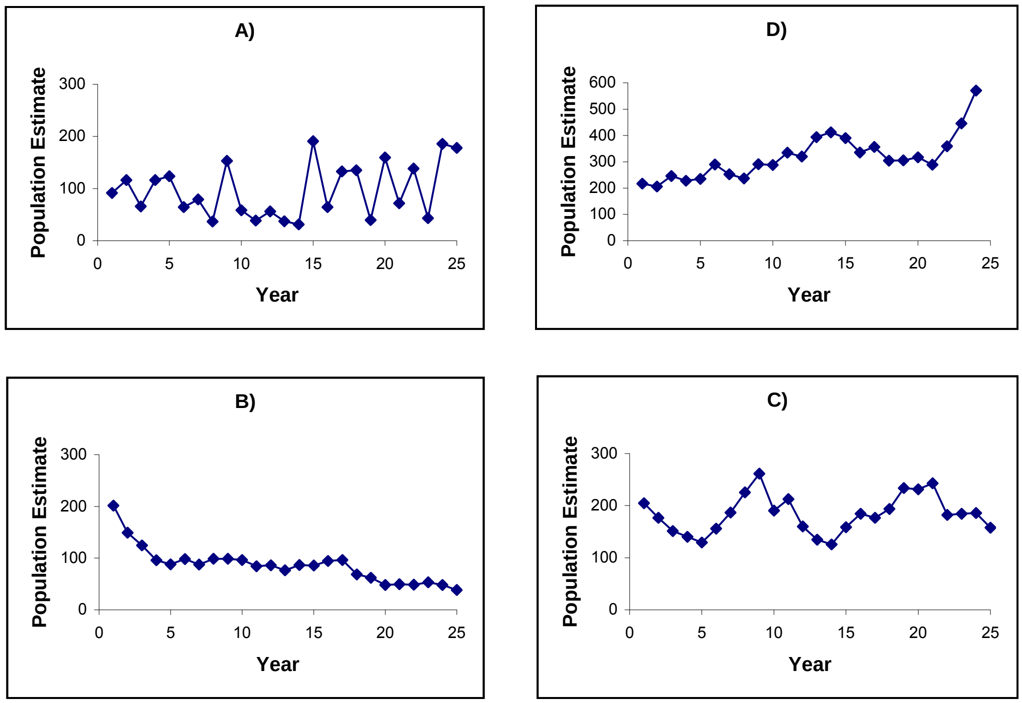


Figure 8. Common trend data likely to be observed during trend monitoring: a) random, b) decreasing, c) increasing, and d) cyclic. Plots based on starting population size of 200 individuals with random change between 0.7 and 1.3 per year. Adapted from Thompson et al. (1998) Figure 5.1, p. 147.

<u>We strongly recommend that survey coordinators review historic time series or literature to estimate the type of trend data the population they are studying is likely to produce before designing a monitoring study (Thompson et al. 1998).</u>

Sources of Variation

There are numerous ways of categorizing variance associated with trend surveys and we recommend the reader review Larsen et al. (2001) and Urquhart et al. (1998) for an in-depth review of the topic. There are more sources of variance when monitoring multiple sites compared to single sites (Larsen et al. 2001). Single sites can have within year and year to year variance. For example, if multiple measurements are taken at a single site within the same year the counts are not likely to be identical each time due to enumeration error and movement in and out of the sample unit (emigration, immigration, births, and deaths). At the same site from year to year there will also likely be a different number counted even if there are no trends present for similar reasons as the within year variance. Multiple site trend surveys can have the same sources of variance as single sites, plus synchronous year to year variance across all sites together (i.e. due to drought, vegetation succession, etc.) and average year to year independent variance at each site (also called interaction or ephemeral spatial and/or temporal variance) (Larsen et al. 2001). Year to year variance at all sites together (coherent variance) and each site independently (interaction variance) has the strongest influence on the ability of a survey design to detect trend and if the variance is relatively large the addition of more samples sites or more revisits will not significantly help (Larsen et al. 2001, Thompson et al. 1998, Urquhart et al 1998). Large year to year variance will require trend surveys to last > 15-20 years to detect modest trends (e.g. 2% per year) (Larsen et al. 2001).

The most effective strategies for dealing with large year to year variance are to stratify sample units and/or populations into groups with similar variance structures (Larsen et al. 2001, Thompson et al. 1998). Stratification of sample units into biologically meaningfull strata (e.g. riffles and pools, lower versus upper reaches, tributary versus mainstem, etc.) can greatly reduce the variance in estimates of distribution and abundance. Stratification could also be done within the target population (e.g. monitor adult age class instead of the entire population). Increased variance in annual counts could be a problem if the survey does not distinguish between groups of individuals that may be influenced by different biological, climatic, and human stressors. For example, in salmonid populations there is a potential to count several different stocks on the spawning grounds, when only one stock is the target of monitoring. Another example could be the potential difficulty in distinguishing between resident and migratory populations of the same species during counts.

Interpreting Trend

If a trend is detected (or not) there are four possible causes: a true trend exists, an intervention has taken place (e.g. severe drought), autocorrelation, or sampling error (Thomas 1996). Section 6.4 discusses the statistical analysis required to determine if a trend is statistical significant. True trend may be influenced by interventions like drought and other extreme environmental conditions and if these events take place during a monitoring program, they will likely bias the results of any trend analysis. Autocorrelation is a common issue in trend analysis because the state of the population in the current year is somewhat dependent on the state of the population in the previous year (Thompson et al. 1998). Autocorrelation can cause an over estimation of power and bias standard errors and confidence intervals. Finally, all count data will have sampling error; however, with proper procedures this should be minimal.

6.1 Objectives

Trend monitoring by definition is interested in a change in a population relative to the existing situation (Elzinga et al. 1998; Thompson et a. 1998). The broad objectives of trend surveys are to monitor change over a specified time. Change/trend objectives are useful when the rate a change a population may be experiencing is important to determine. An appropriate response(s) to the outcome of trend monitoring should be well defined prior to the start of a survey (see Section 2.3.2). Below we outline the management objectives, management responses, and sample objectives for trend surveys. Section 6.1.4 provides some examples of common objectives and responses in fisheries management.

6.1.1 Management Objectives for Trend Surveys

Elzinga et al. (1998) describes management objectives that are focused on a "… change relative to the existing condition" as change/trend objectives. Change/trend objectives are appropriate when a specific future condition can not be clearly defined, but you have an idea of what the rate of change should be (Elzinga et al. 1998). For example, a change/trend objective would be appropriate if you are monitoring a species of management concern for which there is minimal information on historic abundance, but you want to prevent a downward trend from the current abundance. Change/trend management objectives are also used when changes in management have occurred and the response needs to be documented. For example, if some type of habitat restoration has been conducted, there could be an increase, decrease, or random response to the management that you want to track over time.

6.1.2 Management Responses for Trend Surveys

Management responses for trend surveys entail plans to decrease, increase, or maintain the target population. For example, if trend results for a species of management concern demonstrate it is decreasing steadily within an area it was historically abundant, the appropriate management response would be to implement management actions that would promote change in the trend from negative to positive (e.g. barrier or invasive species removal, habitat restoration). Management responses should be clearly stated prior to monitoring.

6.1.3 Sampling Objectives for Trend Surveys

The following sections describe the recommended levels of precision and power, trigger point, and scope if inference that should be used when designing trend surveys. We recognize that specific projects may have site specific objectives that will override these recommendations.

Precision and Power Levels for Trend Surveys

We recognize that some stream systems and species groups may be, for all intents and purposes, impossible to survey for reliable, unbiased fish trend estimates based on available techniques and our current understanding of the stream dynamics. As an example, in sand bed dominated streams of the Coastal

Plains, Adams et al. (2004) found numerous species that varied dramatically over time and space with no apparent habitat associations. In these systems precise abundance estimates for individual species are likely not feasible and other measures such as guild analysis may be more appropriate (S. Adams, Pers. Comm.).

For other less specious and dynamic streams the desired level of precision will vary among projects, but we recommend that trend surveys should have a minimum sample size sufficient to detect a 20% change between sample periods or the duration of the monitoring period 80% of the time with $\alpha = 0.10$. This level of precision is a compromise between the cost of sampling more sites to increase precision and a recognition that populations of fish have an inherent natural variability which can confound our ability to detect changes over time.

Trigger Point for Trend Surveys

The setting of trigger points (where management action is initiated) will depend on the status of the target species, laws and regulations, and public involvement. Study designs should be developed to detect either an increase or decrease, but not both, as it is more efficient and cost effective (i.e. less sample sites required) for one-tailed (i.e. directional) survey designs.

However, as a general rule and for consistency, management actions should be considered when an estimated 20% change in the target population is observed (Vesely et al. 2006).

Scope of Inference for Trend Surveys

The results of any survey can only be applied to areas outside the specific sample points if some type of probabilistic sample design was used (Thompson et al. 1998). When planning trend surveys biologists should try to coordinate their sampling with other agencies and adjacent forests to increase efficiency and their ability to apply the results to larger areas. Recognizing that inventories are likely an ongoing activity, it will be more efficient to plan sampling over the long-term instead of directing sampling at local areas to deal with issues of limited scope.

An example of how a survey design can be expanded to increase its' over all scope and expand the area of statistical inference would be the following:

A watershed with three levels of management, Bureau of Land Management (BLM) manages lower reach, Power Company manages mid reach and reservoir, USFS manages upper watershed. If a probabilistic survey design was implemented throughout the entire watershed to determine trends of species X, it would be more efficient than each agency conducting its own trend survey by reach.

We recommend that randomized survey designs be employed whenever possible and activities be coordinated with other agencies and organizations to increase the efficiency and scope of trend surveys.

6.1.4 Examples of Trend Objectives

The following list provides examples of management objectives, management responses, and sampling objectives for the common types of trend surveys likely to be conducted on National Forests.

Example 1

Management Objective – Allow a decrease of no more than 20% of a population of adult (> 100 mm) smelly darters within the Slimy Creek watershed between 2008 and 2010.

Management Response – If the population estimate of smelly darters decreases by more than 20% by 2010 initiate the restoration efforts outlined in the smelly darter management plan in the summer of 2010.

Sampling Objective – Be 80% sure of detecting a 20% decrease in the smelly darter population with $\alpha = 0.10$.

Example 2

Management Objective – Increase the frequency of occurrence of smelly darters within the Happy Forest Region (populations defined as the presence of adult smelly daters within a subwatershed) by 20% from 2007 to 2015.

Management Response – If the frequency of populations of smelly darters does not increase by 20% by 2015 implement the smelly darter management plan restoration efforts in priority streams.

Sampling Objective - Be 80% sure of detecting a 20% decrease in the smelly darter population with $\alpha = 0.10$.

6.2 Population Measures for Trend Surveys

A wide variety of population and community measures can be used when the objective is to monitor trend. Distribution surveys can provide either presence/absence measures or range measures. Both of these types of distribution measures can be used to detect changes in populations (Holthausen et al. 2005). Presence/absence surveys over time can determine if the frequency of occurrence of a species is changing and range surveys can highlight if the species spatial distribution is expanding, contracting, or staying constant. There is still considerable debate over whether population estimates or relative abundance indices are more suitable for monitoring changes in absolute population density (reviewed in Section 4.0). The decision on whether to use a population estimate or index of abundance to monitor trend will have to made on a case by case basis.

6.3 Field Techniques for Trend Surveys

A discussion of the field techniques used to determine distribution and abundance have been presented in sections 3.0 and 4.0 respectively. The type of field technique used for trend monitoring will depend on the specific objective of the survey. For example, if a precise estimate of change in absolute density of a species is required, then a population estimate technique such as mark-recapture or depletion will be required (Section 4.3.1). Relative abundance measures can be used if less precise estimates of trend are required, but calibration tests of the relative abundance measures should still be conducted annually.

One survey method that is specific to trend monitoring however, is whether to use permanent sample sites or new sites each year (Roper et al. 2003, Larsen et al. 2004). Recent reviews of sampling designs have demonstrated that using permanent sample sites can increase the power to detect trends (Urquhart et al. 1998, Elzinga et al. 2001, Larsen et al. 2001). Using permanent sites will generally decrease the total variance because measurements at the same site will tend to be correlated (Larsen 2001). A substantial reduction in the number of sample sites required to detect trend can be realized using permanent sites since variance is one of the key variables that determine sample size requirements. However, the value of permanent sites is lost if there is little correlation between measured values at a site between consecutive years (Elzinga et al. 1998, Elzinga et al. 2001). The variance of populations of a variety of plants and animals was assessed by Gibbs et al. (1998) and in general, populations that inhabit relatively stable environments or larger, long lived species tended to have lower annual variability. Gibbs et al. (1998) measured variability of populations using the mean coefficient of variation (CV) defined as:

$$CV = SD/X \qquad \text{(Equation 2)}$$

where CV = coefficient of variation, SD = standard deviation, and X = mean.

Salmonid fishes in the studies Gibbs et al. (1998) reviewed had intermediate variability (mean CV = 47%) whereas non-salmonid fishes had high variability (mean CV = 71%). These estimates were based on a review of 42 salmonid and 30 non-salmonid count series of at least 5 years.

There are a variety of reasons other than population variability that can lead to limited correlation between counts at permanent sites including observer error and variable application of sampling protocols, different sampling times, and imprecise location of permanent sites (Larsen et al. 2001). Very concise sampling protocols and substantial field crew training can reduce some of these sources of error (Roper et al. 2002).

There are several potential disadvantages of permanent sample sites such as (Elzinga et al. 1998, Elzinga et al. 2001, Roper et al. 2003):

- Cost more to set up
- Can be difficult to relocate accurately

- Frequent return visits can disrupt the site physically or the target population (i.e. injury to fish from repeated electrofishing and handling)
- Depend on relatively high correlation of measurements between years
- Are more suited to long lived, relatively stable populations, and
- Have a reduced ability to describe status.

We recommend that permanent sample sites be established for most trend monitoring. Situations where permanent sites would not be appropriate include streams where there is little correlation between abundance and time. Examples of these types of streams may include sand bed streams with highly mobile stream channels, and streams that frequently dewater.

When population estimates are being used for trend monitoring sampling should be done using block nets at the upstream and downstream ends of all sample units.

6.4 Data Analysis for Trend Surveys

Elzinga et al. (1998, 2001) has an extensive review of the data analysis techniques used for trend analysis and we summarize the key findings below. Regression analysis is the most commonly used data analysis technique for trend data whereby the count (population estimate or indices) is plotted on the y axis and time (usually year) is plotted on the x axis. A minimum of five years of data are recommend to accurately determine if a trend exists. A regression equation can be calculated with most spreadsheet and statistical packages including EXCEL, JMP, SAS, and R. The resulting equation will have two parameter estimates, one for the slope of the regression line and one for the y intercept (Figure 9). The slope of the line indicates whether there is a decrease, increase, or stable population from year to year. The p-value associated with the regression analysis indicates if the trend is significantly different from "0" or no trend. Researchers should specify the appropriate level of α prior to initiating the study.

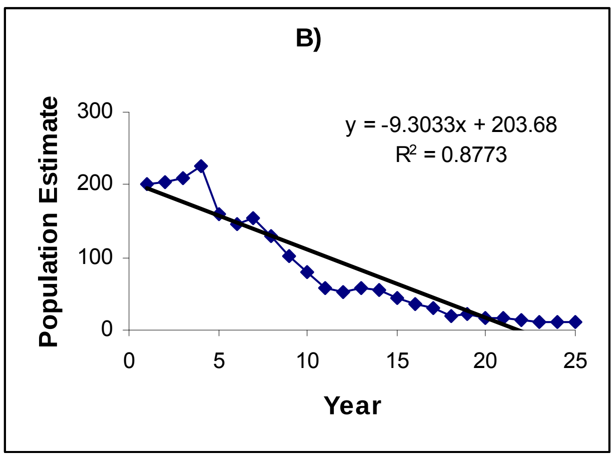


Figure 9. An example of regression analysis for detecting trend.

Many populations will exhibit an exponential change over time (e.g. constant rate of 2%) rather than a constant change in the number of individuals (e.g. decrease of 100 individuals each year) (Elzinga et al. 1998, Elzinga et al. 2001, Gibbs and de Arellano 2007). Regression analysis assumes a linear, or constant change over time and equal variances between years. Because trend data is often not linear and can have changing variability between years the data are usually transformed using some type of log transformation. Elzinga et al. (1998, 2001) describe several other alternatives to analyzing trend data with other parametric tests and nonparametric tests.

6.5 Analysis Tools for Trend Surveys

There are a variety of tools for determining the sample size required for trend surveys based on either a single point or multiple sites. Three of the most commonly used programs are MONITOR (Gibbs and de Arellano 2007), PRESENCE (Mackenzie et al. 2002), and TRENDS (Gerrodette 1993). Spreadsheet programs and statistical software for trend analysis are reviewed in Vesely et al. (2006) and Appendix 7 provides links to statistical software and documentation.

Chapter 7.0 Strategies for Purposive Sampling

Many of the current and past fish surveys were initiated to address site specific objectives related to enhancement or restoration projects at the reach, valley segment, or sub-basin scale (Dolloff et al. 1997). These efforts have provided valuable information on local fisheries resources. However, differences in approaches and protocols have made it difficult to integrate and analyze data across lands managed by the Forest Service. This inability to integrate data has made it difficult for the agency to describe the status of fish populations and habitat conditions at scales of interest to regulatory agencies, the public, and national forest planning efforts.

Although this manual was developed primarily for large scale surveys, we recognize that local interest sometimes supersede these objectives. The following chapter briefly outlines the limitations of purposive sampling and when it is appropriate.

7.1 Appropriate Situations for Purposive Sampling

The potential number of project specific surveys undertaken throughout the National Forest lands are extremely large, but they all have one thing in common: their objective(s) are usually limited in scope. Examples of such projects include: i) fixing a perched culvert and monitoring the impact of fish species distribution, ii) adding large woody debris (LWD) to a reach and monitoring the affect on the abundance of species X, and iii) annually resampling representative sites for species X to compare present abundance with historic abundance. These are appropriate situations to use purposive sampling provided it is explicitly recognized what information these types of studies can and can not provide. For example, sampling near bridge crossings to assess whether a species distribution has contracted from historic levels will not allow managers to draw statistically valid conclusions about a species population in other parts of the watershed.

One of the largest strengths of purposive sampling can be in rapid field assessments that will help managers develop hypotheses that can tested more rigorously with formal statistical surveys. For example, a new road may be proposed for a particular watershed and there are two route options that the road can take: along stream 1 or stream 2. There is only very old distribution data that suggests species X is present in both streams. Before developing a large scale survey design for the two streams, informal surveys could be conducted to see if there are potential differences in the populations and habitat types within the two streams. Informal surveys could include walking the banks to detect fish visually, angling surveys, minnow traps, or electrofishing surveys without block nets. Results from any of these methods would provide some idea of the distribution or abundance of species X within the two streams, but they would not provide any measure of precision or confidence. Informal studies could also indicate if there are some obvious differences between the two streams (i.e. channel morphology, habitat complexity, species richness, etc.) and help managers begin to plan a more formal survey design to determine population estimates of species X in each stream.

The other advantage of purposive sampling is that if the presence of a particular species is all that needs to be established, and it is established during the informal survey process, time and money can be saved compared to conducting a formal statistical survey. Of course if the species is not detected during the informal survey then a formal survey would be required to determine true absence at a specific threshold density.

Although the management objectives and responses for purposive sampling will be varied, we recommend that the sampling objectives outlined in this guide still be used whenever possible. We recommend that all fish surveys be designed to estimate the frequency or abundance of a species within 20% of the true abundance 80% of the time with $\alpha = 0.10$. A trigger point of 20% change should be used in most cases unless local conditions dictate otherwise. Study designs should be designed to detect a one-way change (either up or down) which will require fewer sample sites because it employs one-tailed instead of two-tailed tests (Vesely et al. 2006).

7.2 Inappropriate Situations for Purposive Sampling

It should be assumed more often than not that any individual inventory project will likely not be the last study in a particular stream. Therefore, if the distribution of a particular fish species is required on a regular basis for different streams within a larger watershed it is more cost effective to conduct one large scale presence/absence survey than to conduct numerous small scale surveys. If the purpose of the survey is to extrapolate the results to a broader area than the sample sites were originally chosen from (or if the samples were not randomly selected) then it would be inappropriate to conduct a purposive survey.

Finally, an informal survey based on a non-randomized design is not appropriate for determining the current status of a species, or to monitor trends over time. Only the rigorous survey design protocols such as those described in this guide for population estimation (section 5.2.1) or trend surveys (section 6.2) will provide estimates with measurable levels of precision and confidence.

Appendix 1. Glossary

Alpha value (α)	- Type I error rate; probability of rejecting the null hypothesis when it is true; also a value used to set the level of confidence in a confidence interval
Beta value (ß)	- Type II error rate; the probability of failing to reject the null hypothesis when it is false; Power = 1-ß
Closed Population	- assumed no births, deaths, immigration, or emigration within a specified area during a specified time
Confidence interval	- an interval around a paramter estimate that provides a measure of confidence regarding how close a sample based estimate is to the true parameter
Element	- an individual fish, object, or item that is measured, counted, or recorded
Index Count	- a relative measure assumed to be correlated to the true parameter; any partial count that is not adjusted for capture efficiency
Nonrandom Sample	- subjectively choosing sample sites and units
P value	- in hypothesis testing, the probability that an observed difference between the intervention and control groups is due to chance alone if the null hypothesis is true.
Power (1-ß)	- the probability to detect a statistically significant difference in a test of the null hypothesis given the difference is present
Precision	- the degree of spread in estimates generated from repeated samples
Random Sample	- a collection of sample units chosen based on some known chance of selection
Sample frame	- all the possible sample units within an area of study
Sample unit	- the specific area where an individual sample is collected; this guide recommends 100 m stream segments
Stream Order –	- this manual uses Strahler's (1957) system to describe stream order. The smallest streams on a map are considered 1st order streams. The confluence (joining) of two 1st order streams forms a 2nd order stream; the confluence of two 2nd order streams forms a 3rd order stream. Stream order (stream size) affects a stream's natural characteristics, including the biological communities that live in the stream, such as fish and invertebrates. Many 1st order streams are intermittent (do not have continuous year round water flow). Stream order is scale dependant and usually 1:50,000 or 1:24,000 maps are used to determine stream order.
Target population	- all the elements within a sample frame during a specific time
Type I Error	- rejection of the null hypothesis when it is true
Type II Error	- accpetance of the null hypothesis when it is false
Variance	- a statistical measure of precision
Wadeable stream	- a small stream that can be sampled without the use of a boat and that generally between a 1st and 5th order stream

Appendix 2. A sample of reported capture efficiencies for common species and sample techniques.

Species/ Group	Age	Size (mm)	Sample Technique[a]	State/ Stream Type	Approx. CE (%)[b]	Reference/ Comments
Bull trout		70-200	DS NS	WA, Cold	12-14 30-40	Peterson et al. (2002)
Bull trout		70-200	EL & DS	ID, Cold	25*	Rieman and McIntyre (1995) * assumed not measured
Brook,Brown trout	1		2 pass EL	CO, cold	19-84	Riley and Fausch (1992)
Brook,Brown, Rainbow trout	>1				66-84	
Brook,Brown trout	1		3 pass EL		35-84	
Brook,Brown, Rainbow trout	>1				52-84	
Brook Trout	0 ≥ 1		3 pass EL	WY, cold	42-83 92-96	Thompson and Rahel (1996)
				TN		Habera et al. (1992)
Westslope Cutthroat Trout		100-199	1 pass EL ≥ 2 pass EL		25-30 5-15	Peterson et al. (2004)
Bull Trout		70-99 100-199	1 pass EL ≥ 2 pass EL		7.5-15 3-6	Peterson et al. (2004)
Cyprinidae Cottidae Percidae Ictaluridae		< 150	QS	Ozark	84 80 57 31	Peterson and Rabeni (2001)
Bluegill, Gizzard Shad, Common Carp, River Carp Sucker, and others	-	-	EL(Boat)	OK,	68-100	Layher and Maughan (1984)

[a] DS = day time snorkel, EL = backpack electrofishnig, NS = night snorkel, QS = quadrat sampler.
[b] Approx. CE(%) = approximate capture efficiency percent (i.e. the estimated percent of the true number of fish within a sample unit captured by each technique).

Appendix 3. Methods for calculating sample size based on the objectives of the study (reproduced from Elzinga et al. 1998).

We reproduce three different sample size equations in this appendix for determining as per Elzinga et al. (1998):

A) the necessary sample size for estimating a single population mean or a single population total with a specified level of precision,

B) the necessary sample size for detecting differences between two means when using paired or permanent sampling units, and

C) the necessary sample size for estimating a single population proportion with a specified level of precision.

Elzinga et al. (1998) provides these equations, plus equations for determining the necessary sample size for detecting differences between two means with temporary sampling units, and the necessary sample size for detecting differences between two proportions with temporary sampling units.

Each section below includes the sample size equation, a description of each term in the equation, a table of appropriate coefficients, and a worked out example based on a stated management and sampling objective. The examples included in this appendix all refer to monitoring with a 100 m sample unit (reach). The equations and calculations also work with other kinds of monitoring data such as measurements based on quadrat sampling. The sampling objectives and worked-out examples show calculations for two-tailed significance tests. This implies an interest in being able to detect either *increases* or *decreases* over time, even though the management objectives specify a desire to achieve a change in only one direction or the other. If you are only interested in detecting changes in one direction, and you only plan on analyzing your monitoring results with one directional null hypotheses (e.g., Ho = density has not increased), then you should apply a simple modification to the simple size procedures. To change any sample size procedure to a one-tailed situation, simply double the false-change (Type I) error rate (α) and look up the new doubled-α value in the table of coefficients (e.g., use α = 0.20 instead of α = 0.10 for a one-tailed test with a false-change (Type I) error rate of α = 0.10).

The coefficients used in all of the equations are from a standard normal distribution (Z_α and Z_β) instead of the t-distribution (t_α and t_β). These two distributions are nearly identical at large sample sizes, but at small sample sizes (n < 30) the Z coefficients will slightly underestimate the number of sampling units needed. The correction procedure described for situation A) already adjusts the sample size using the appropriate *t*-value. For the other equations, t_α and t_β values can be obtained from a *t*-table and used in place of the Z_α and Z_β coefficients that are included with the sample size equations (see any standard statistics textbook for a *t*-table). The appropriate t_α-coefficient for the false-change (Type I) error rate can be taken directly from the α(2) column of a *t*-table at the appropriate degrees of freedom (*v*). For example, for a false-change error rate of 0.10 use the α(2) = 0.10 column. The appropriate t_β coefficient for a specified missed-change error level can be looked up by calculating 2(1-power) and looking up that value in the appropriate α(2) column. For example, for a power of 0.90, the calculations for t_β would be 2(1-.90) = 0.20. Use the α(2) = 0.20 column at the appropriate degrees of freedom (*v*) to obtain the appropriate t_β value.

A) Sample size equation for determining the necessary sample size for estimating a single population mean or a population total with a specified level of precision.

Estimating a sample *mean vs. total population size*. The sample size needed to estimate confidence intervals that are within a given percentage of the estimated *total population size* is the same as the sample size needed to estimate confidence intervals that are within that percentage of the estimated *mean value*. The instructions below assume you are working with a sample mean. Determining sample size for a single population mean or a single population total is a two- or three-step process.

(1) The first step is to use the equation provided below to calculate an uncorrected sample size estimate.

(2) The second step is to consult the Sample Size Correction Table (below) to come up with the corrected sample size estimate. The use of the correction table is necessary because the equation below under-estimates the number of sampling units that will be needed to meet the specified level of precision. The use of the table to correct the underestimated sample size is simpler than using a more complex equation that does not require correction.

(3) The third step is to multiply the corrected sample size estimate by the finite population correction factor (FPC) if more than 5% of the population area is being sampled.

1. Calculate an initial sample size using the following equation:

$$n = \frac{(Z_\alpha)^2 (s)^2}{(B)^2}$$

Where:

n = uncorrected sample size estimate.

Z_α = standard normal coefficient from the table below.

s = standard deviation.

B = desired precision level expressed as half of the maximum acceptable confidence interval width. This needs to be specified in absolute terms rather than as a percentage. For example, if you wanted your confidence interval width to be within 30% of your sample mean (i.e., $\bar{x}$ ± 30% * $\bar{x}$) and your sample mean = 10 fish/sample unit then B = (0.30 x 10) = 3.0.

Standard normal deviates (Z_α) for various confidence levels		
Confidence level (%)	Alpha (α) level	(Z_α)
80	0.20	1.28
90	0.10	1.64
95	0.05	1.96
99	0.01	2.58

2. To obtain the adjusted sample size estimate, consult the Sample Size Correction Table below.

n = the uncorrected sample size value from the sample size equation.

n* = the corrected sample size value.

3. Additional correction for sampling finite populations.

The above formula assumes that the population is very large compared to the proportion of the population that is sampled. If you are sampling more than 5% of the whole population then you should apply a correction to the sample size estimate that incorporates the finite population correction (FPC) factor. This will reduce the sample size.

The formula for correcting the sample size estimate with the FPC for confidence intervals is:

$$n' = \frac{n^*}{(1 + (n^* / N))}$$

Where:

n' = The new FPC-corrected sample size.

n* = The corrected sample size from the sample size correction

table (Table 1).

N = The total number of possible sample unit locations in the population. To calculate N, determine the total area of the population and divide by the size of one sample unit .

Sample Size Correction Table.

80% confidence level						90% confidence level					
n	n*	n	n*	n	n*	n	n*	n	n*	n	n*
1	5	51	65	101	120	1	5	51	65	101	120
2	6	52	66	102	121	2	6	52	66	102	122
3	7	53	67	103	122	3	8	53	67	103	123
4	9	54	68	104	123	4	9	54	69	104	124
5	10	55	69	105	124	5	11	55	70	105	125
6	11	56	70	106	125	6	12	56	71	106	126
7	13	57	71	107	126	7	13	57	72	107	127
8	14	58	73	108	128	8	15	58	73	108	128
9	15	59	74	109	129	9	16	59	74	109	129
10	17	60	75	110	130	10	17	60	75	110	130
11	18	61	76	111	131	11	18	61	76	111	131
12	19	62	77	112	132	12	20	62	78	112	132
13	20	63	78	113	133	13	21	63	79	113	134
14	22	64	79	114	134	14	22	64	80	114	135
15	23	65	80	115	135	15	23	65	81	115	136
16	24	66	82	116	136	16	25	66	82	116	137
17	25	67	83	117	137	17	26	67	83	117	138
18	27	68	84	118	138	18	27	68	84	118	139
19	28	69	85	119	140	19	28	69	85	119	140
20	29	70	86	120	141	20	29	70	86	120	141
21	30	71	87	121	142	21	31	71	88	121	142
22	31	72	88	122	143	22	32	72	89	122	143
23	33	73	89	123	144	23	33	73	90	123	144
24	34	74	90	124	145	24	34	74	91	124	145
25	35	75	91	125	146	25	35	75	92	125	147
26	36	76	93	126	147	26	37	76	93	126	148
27	37	77	94	127	148	27	38	77	94	127	149
28	38	78	95	128	149	28	39	78	95	128	150
29	40	79	96	129	150	29	40	79	96	129	151
30	41	80	97	130	151	30	41	80	97	130	152
31	42	81	98	131	152	31	42	81	99	131	153
32	43	82	99	132	154	32	44	82	100	132	154
33	44	83	100	133	155	33	45	83	101	133	155
34	45	84	101	134	156	34	46	84	102	134	156
35	47	85	102	135	157	35	47	85	103	135	157
36	48	86	104	136	158	36	48	86	104	136	158
37	49	87	105	137	159	37	49	87	105	137	159
38	50	88	106	138	160	38	50	88	106	138	161
39	51	89	107	139	161	39	52	89	107	139	162
40	52	90	108	140	162	40	53	90	108	140	163
41	53	91	109	141	163	41	54	91	110	141	164
42	55	92	110	142	164	42	55	92	111	142	165
43	56	93	111	143	165	43	56	93	112	143	166
44	57	94	112	144	166	44	57	94	113	144	167
45	58	95	113	145	168	45	58	95	114	145	168
46	59	96	115	146	169	46	60	96	115	146	169
47	60	97	116	147	170	47	61	97	116	147	170
48	61	98	117	148	171	48	62	98	117	148	171
49	62	99	118	149	172	49	63	99	118	149	172
50	64	100	119	150	173	50	64	100	119	150	173

Sample Size Correction Table con't.

95% confidence level						99% confidence level					
n	n*	n	n*	n	n*	n	n*	n	n*	n	n*
1	5	51	66	101	121	1	6	51	67	101	122
2	7	52	67	102	122	2	8	52	68	102	123
3	8	53	68	103	123	3	9	53	69	103	124
4	10	54	69	104	124	4	11	54	70	104	126
5	11	55	70	105	125	5	12	55	72	105	127
6	12	56	71	106	126	6	14	56	73	106	128
7	14	57	72	107	128	7	15	57	74	107	129
8	15	58	74	108	129	8	16	58	75	108	130
9	16	59	75	109	130	9	18	59	76	109	131
10	18	60	76	110	131	10	19	60	77	110	132
11	19	61	77	111	132	11	20	61	78	111	133
12	20	62	78	112	133	12	21	62	79	112	134
13	21	63	79	113	134	13	23	63	80	113	135
14	23	64	80	114	135	14	24	64	82	114	136
15	24	65	81	115	136	15	25	65	83	115	138
16	25	66	83	116	137	16	26	66	84	116	139
17	26	67	84	117	138	17	28	67	85	117	140
18	28	68	85	118	139	18	29	68	86	118	141
19	29	69	86	119	141	19	30	69	87	119	142
20	30	70	87	120	142	20	31	70	88	120	143
21	31	71	88	121	143	21	32	71	89	121	144
22	32	72	89	122	144	22	34	72	90	122	145
23	34	73	90	123	145	23	35	73	92	123	146
24	35	74	91	124	146	24	36	74	93	124	147
25	36	75	92	125	147	25	37	75	94	125	148
26	37	76	94	126	148	26	38	76	95	126	149
27	38	77	95	127	149	27	39	77	96	127	150
28	39	78	96	128	150	28	41	78	97	128	152
29	41	79	97	129	151	29	42	79	98	129	153
30	42	80	98	130	152	30	43	80	99	130	154
31	43	81	99	131	154	31	44	81	100	131	155
32	44	82	100	132	155	32	45	82	101	132	156
33	45	83	101	133	156	33	46	83	103	133	157
34	46	84	102	134	157	34	48	84	104	134	158
35	48	85	103	135	158	35	49	85	105	135	159
36	49	86	105	136	159	36	50	86	106	136	160
37	50	87	106	137	160	37	51	87	107	137	161
38	51	88	107	138	161	38	52	88	108	138	162
39	52	89	108	139	162	39	53	89	109	139	163
40	53	90	109	140	163	40	55	90	110	140	165
41	54	91	110	141	164	41	56	91	111	141	166
42	56	92	111	142	165	42	57	92	112	142	167
43	57	93	112	143	166	43	58	93	114	143	168
44	58	94	113	144	168	44	59	94	115	144	169
45	59	95	114	145	169	45	60	95	116	145	170
46	60	96	116	146	170	46	61	96	117	146	171
47	61	97	117	147	171	47	62	97	118	147	172
48	62	98	118	148	172	48	64	98	119	148	173
49	63	99	119	149	173	49	65	99	120	149	174
50	65	100	120	150	174	50	66	100	121	150	175

Example:

Management objective:

Restore the population of species Y in population Z to a density of at least 30 fish/sample unit by the year 2010.

Sampling objective:

Obtain estimates of the mean density and population size with 80% confidence intervals that are within 20% of the estimated true value.

Results of pilot sampling:

Mean ($\bar{x}$) = 25 fish/sample unit.

Standard deviation (s) = 7 fish.

Given:

The desired **confidence level** is 95% so the appropriate Z_α from the table above = 1.96.

The desired **confidence interval width** is 20% (0.20) of the estimated true value. Since the estimated true value is 25 fish/sample unit, the desired confidence interval (**B**) = 25 x 0.20 = 5 fish/sample unit. Calculate an unadjusted estimate of the sample size needed by using the sample size formula: Round 7.5 plots up to 8 sample units for the unadjusted sample size. To adjust this preliminary estimate, go to the Sample Size Correction Table and find n = 8 and the corresponding n* value in the 95% confidence level portion of the table. For n = 8, the corresponding n* value = 15. The corrected estimated sample size needed to be 95% confident that the estimate of the population mean is within 20% (+/- 5 fish) of the true mean = **15 sample units**.

If the pilot data described above was gathered using a 100 m sample unit and the total population being sampled was located within a 1900 m long stream then N = 1900 m/100 m = 19. The corrected sample size would then be:

$$n' = \frac{n^*}{(1+(n^*/N))} \Rightarrow n' = \frac{15}{(1+(15/19))} = 8.4$$

The new, FPC-corrected, estimated sample size to be 95% confident that the estimate of the population mean is within 20% (+/- 5 fish) of the true mean = **9 sample units**.

B) Sample size equation for determining the necessary sample size for detecting differences between two means when using paired or permanent sampling units.

When paired sampling units are being compared or when data from permanent sample units are being compared between two time periods, then sample size determination requires a different procedure than if samples are independent of one another. The equation for determining the number of samples necessary to detect some "true" difference between two sample means is:

$$n = \frac{(s)^2 (Z_\alpha + Z_\beta)^2}{(MDC)^2}$$

Where:

s = Standard deviation of the differences between paired samples (see examples below).

Z_α = Z-coefficient for the false-change (Type I) error rate from the table below.

Z_β = Z-coefficient for the missed-change (Type II) error rate from the table below.

MDC = Minimum detectable change size. This needs to be specified in absolute terms rather than as a percentage. For example, if you wanted to detect a 20% change in the sample mean from one year to the next and your first year sample mean = 10 fish/sample unit then

MDC = (0.20 x 10) = 2 fish/sample unit.

Table of standard normal deviates for Z_α		Table of standard normal deviates for Z_β		
False-change (Type I) error rate (α)	Z_α	Missed-change (Type II) error rate (β)	Power	Z_β
0.40	0.84	0.40	0.60	0.25
0.20	1.28	0.20	0.80	0.84
0.10	1.64	0.10	0.90	1.28
0.05	1.96	0.05	0.95	1.64
0.01	2.58	0.01	0.99	2.33

If the objective is to track changes over time with permanent sampling units and only a single year of data is available, then you will not have a standard deviation of differences between the paired samples. If you have an estimate of the likely degree of correlation between the two years of data, and you assume that the among sampling units standard deviation is going to be the same in the second time period, then you can use the equation below to estimate the standard deviation of differences.

$$s_{diff} = (s1)\left(\sqrt{(2-(1-corr_{diff}))}\right)$$

Where:

s_{diff} = Estimated standard deviation of the differences between paired samples.

s_1 = Sample standard deviation among sampling units at the first time period.

$corr_{diff}$ = Correlation coefficient between sampling unit values in the first time period and sampling unit values in the second time period.

Example #1:

Management objective:

Achieve at least a 20% higher number of species X per m^2 at site Y in areas with restored riparian vegetation as compared to areas not restored in 1999.

Sampling objective:

To detect a 20% difference in mean fish density in areas with restored riparian vegetation compared to areas without restored riparian vegetation. Want to be 90% certain of detecting that difference, if it occurs, and are willing to accept a 10% chance of making a false-change error (i.e. conclude that a difference exists when it really did not).

Results from pilot sampling:

Five paired sample units were sampled where one member of the pair had riparian restoration and the other member of the pair did not.

Sample unit pair no.	No. fish/sample unit: Restored	No. fish/sample unit: Not restored	Difference between restored and not restored
1	2	3	1
2	5	8	3
3	4	9	5
4	7	12	5
5	3	7	4
	$\bar{x}$ =4.20 s=1.92	$\bar{x}$ =7.80 s=3.27	$\bar{x}$ 3.60 s=1.67

Given:

The sampling objective specified a desired minimum detectable difference (i.e., equivalent to the MDC) of 20%. Taking the larger of the two mean values and multiplying by 20% leads to:

(7.80) x (0.20) = **MDC = 1.56** fish sample unit

The appropriate **standard deviation** to use is **1.67**, the standard deviation of the differences between the pairs. The acceptable **False-change error rate (α) = 0.10**, so the appropriate Z_α from the table = 1.64. The desired Power is 90% (0.90), so the **Missed-change error rate (β) = 0.10** and the appropriate Z_β coefficient from the table = 1.28.

Calculate the estimated necessary sample size using the equation provided above:

$$n = \frac{(s)^2 (Z_\alpha + Z_\beta)^2}{(MDC)^2} \qquad n = \frac{(1.67)^2 (1.64 + 1.28)^2}{(1.56)^2} = 9.7$$

Round up 9.7 to 10 sample units.

Final estimated sample size needed to be 90% certain of detecting a true difference of 1.56 fish/sample unit between the restored and un-restored sample units with a false-change error rate of 0.10 = **10 sample units**.

** Note if you wanted to be able to detect a 20% difference in mean fish between two years the procedure for determining the necessary sample size would be very similar to the previous example. Just replace "restored" and "not restored" in the data table with the two years and the rest of the calculations would be the same. Because the sample size determination procedure needs the standard deviation of the difference between two samples, you will not have the necessary standard deviation term to plug into the equation until you have two years of data. The standard deviation of the difference can be estimated in the first year if some estimate of the correlation coefficient between sampling unit values in the first time period and the sampling unit values in the second time period is available (see the s_{diff} equation above).

Correction for sampling finite populations:

The above formula assumes that the population is very large compared to the proportion of the population that is sampled. If you are sampling more than 5% of the whole population area then you should apply the FPC factor as in section a point 3 above.

Note on the statistical analysis for two sample tests from finite populations.

If you have sampled more than 5% of an entire population then you should also apply the finite population correction factor to the results of the statistical test. This procedure involves dividing the test statistic by the square root of (1-n/N). For example, if your t-statistic from a particular test turned out to be 1.782 and you sampled n=9 sample units out of a total N=50 possible sample units, then your correction procedure would look like the following:

$$t' = \frac{t}{\sqrt{1-(n/N)}} \qquad t' = \frac{1.782}{\sqrt{1-(9/50)}} = 1.968$$

Where:

t = The *t*-statistic from a *t*-test.

t' = The corrected *t*-statistic using the FPC.

n = The sample size from the equation above.

N = The total number of possible sample units in the population. To calculate N, determine the total area of the population and divide by the size of each individual sampling unit.

You would need to look up the *p*-value of *t*' = 1.968 in a *t*-table for the appropriate degrees of freedom to obtain the correct *p*-value for this statistical test.

C) Sample size equation to determining the necessary sample size for estimating a single population proportion with a specified level of precision.

The equation for determining the sample size for estimating a single proportion is:

$$n = \frac{(Z_\alpha)^2 (p)(q)}{d^2}$$

Where:

n = estimated necessary sample size.

Z_α = coefficient from the table of standard normal deviates below.

p = value of the proportion as a decimal percent (e.g., 0.45). If you don't have an estimate of the current proportion, use 0.50 as a conservative estimate.

q = 1 - p.

d = desired precision level expressed as half of the maximum acceptable confidence interval width. This is also expressed as a decimal percent (e.g., 0.15) and this represents an *absolute* rather than a *relative* value. For example, if your proportion value is 30% and you want a precision level of ±10% this means you are targeting an interval width from 20% to 40%. Use 0.10 for the d-value and *not* 0.30 x 0.10 = 0.03. See the Table of standard normal deviates (Z_α) in section A) point 1 for various confidence levels.

Example:

Management objective:

Maintain at least a 40% frequency (in 100 m sample units) of species Y in population Z over the next 5 years.

Sampling objective:

Estimate percent frequency with 95% confidence intervals no wider than ± 10% of the estimated true value.

Results of pilot sampling:

The proportion of sample units with species Z is estimated to be p = 65% (0.65). Because q = (1-p), q = (1-0.65) = 0.35.

Given:

The desired **confidence level** is 95% so the appropriate Z_α from the table above = 1.96.

The desired **confidence interval width (d)** is specified as 10% (0.10). Using the equation provided above:

$$n = \frac{(Z_\alpha)^2 (p)(q)}{d^2} \qquad n = \frac{(1.96)^2 (0.65)(0.35)}{0.10^2} = 87.4$$

Round up 87.4 to 88.

The estimated sample size needed to be 95% confident that the estimate of the population percent frequency is within 10% (+/- 0.10) of the true percent frequency = **88 sample units**.

This sample size formula works well as long as the proportion is more than 0.20 and less than 0.80 (Zar 1984). If you suspect the population proportion is less than 0.20 or greater than 0.80, use 0.20 or 0.80, respectively, as a conservative estimate of the proportion.

Correction for sampling finite populations:

The above formula assumes that the population is very large compared to the proportion of the population that is sampled. If you are sampling more than 5% of the whole population area then you should apply a correction for your sample size estimate that incorporates the FPC factor as above.

Appendix 4. Random number table.

0.405495	0.813392	0.852563	0.155490	0.818997	0.400879	0.903331	0.953275	0.763381	0.169939
0.318666	0.540615	0.289456	0.201762	0.171511	0.974278	0.292963	0.381072	0.452560	0.178461
0.895380	0.600386	0.734937	0.887684	0.532542	0.085199	0.142989	0.435226	0.841223	0.767137
0.872020	0.781283	0.851746	0.080955	0.120079	0.431049	0.568847	0.497585	0.729937	0.455548
0.011311	0.117469	0.542646	0.945093	0.843107	0.812041	0.412282	0.080605	0.140443	0.821235
0.894534	0.222384	0.454479	0.747548	0.181953	0.637144	0.559805	0.986027	0.223623	0.079262
0.017626	0.774187	0.180326	0.218857	0.802813	0.157827	0.352038	0.535237	0.135093	0.942404
0.015399	0.647913	0.111685	0.924481	0.077806	0.229232	0.840082	0.511766	0.128777	0.889048
0.462705	0.935760	0.379101	0.611709	0.091419	0.394186	0.158498	0.117643	0.188914	0.367009
0.681262	0.333324	0.963501	0.600160	0.777600	0.579064	0.611118	0.273477	0.667802	0.388441
0.986447	0.568944	0.685957	0.486091	0.707127	0.520814	0.328147	0.748553	0.103651	0.056035
0.834966	0.615647	0.629959	0.768745	0.630382	0.876066	0.906841	0.285624	0.657621	0.460327
0.662463	0.059736	0.579907	0.137643	0.450397	0.237314	0.195952	0.112555	0.996775	0.974260
0.208825	0.721738	0.918331	0.129124	0.623189	0.033686	0.223322	0.033340	0.173632	0.632154
0.406315	0.200167	0.120294	0.284933	0.557224	0.557761	0.418773	0.533717	0.531488	0.916887
0.240670	0.844530	0.013752	0.856453	0.730686	0.788203	0.728948	0.465120	0.698776	0.159954
0.081097	0.738048	0.824249	0.560537	0.438832	0.985607	0.787313	0.564606	0.695935	0.181642
0.683667	0.178841	0.624009	0.329889	0.711999	0.301693	0.381480	0.825740	0.798090	0.035639
0.285602	0.526550	0.813638	0.123819	0.809829	0.569598	0.359208	0.119089	0.052269	0.178827
0.044242	0.999734	0.067530	0.599051	0.185705	0.953915	0.234311	0.136284	0.398569	0.131006
0.874275	0.192581	0.978862	0.354260	0.576068	0.874024	0.245809	0.355408	0.872372	0.637927
0.811739	0.652871	0.640349	0.061588	0.039026	0.134228	0.309270	0.985289	0.857977	0.424607
0.279505	0.955976	0.549045	0.194253	0.987313	0.446818	0.506125	0.693791	0.296294	0.633898
0.311933	0.170232	0.501312	0.612332	0.788395	0.172284	0.514956	0.551749	0.660397	0.676089
0.110531	0.682404	0.326738	0.966555	0.605252	0.181531	0.926422	0.146537	0.912571	0.609319

Appendix 5. Fish survey protocols and applications for electrofishnig, seines, plot/quadrat, underwater (snorkel), redd count, and minnow trap techniques .

The following safety procedures, equipment checks, and survey techniques are summarized primarily from Murphy and Willis (1996), Resource Inventory Committee (1997), Bonar et al. (In Press), and Johnson et al. (2007) fish survey manuals. The following are point form summaries of the recommend techniques for the most common survey techniques for wadeable streams regardless of the survey type. We have attempted to incorporate all the details of the original authors in a concise format. Please review the cited manuals for more complete descriptions of the techniques and rationale or their use. The survey techniques are divided into two groups A) intrusive techniques (fish are captured with an active method), and B) non-intrusive methods (fish are either observed or captured passively and with a low probability of injury).

Safety Considerations and Equipment Check

Prior to Formal Sampling (** applies to electrofishing only)

- Submit an activity plan and emergency procedures with office staff or designated person
- Use a minimum of two people per crew in any field situation
- All crew members should have completed a training course (## recommended)

- Go over safety and operating procedures with crew members
- Equipment check (dive, gear, holes in nets, terminals, connections, battery charged, gloves, etc.)
- Test equipment in a known fish bearing area outside the sample site
- **Optimize settings for fish capture without injuring fish
    - o Recommend direct DC for threatened on endangered species and pulsed DC for all other species
    - o Start with low settings: voltage (100-200 vol.), frequency (30 Hz), and pulse width, also known as duty cycle (4-5 ms)
    - o Increase voltage first if fish are not properly immobilized in 100 volt increments up to max 1100 volts
    - o If fish are still not properly immobilized, decrease voltage to 300 volts and increase frequency by 10-15 Hz increments until desired results are reached
    - o Continually examine fish for injury and adjust settings as needed
- **Measure conductivity and stream temperature
    - o Conductivity below (< 100 µmhos/cm) will require salt blocks placed upstream of the sample unit
    - o Temperatures < 4 C should not be sampled because capture probabilities are significantly reduced due to fish inactivity

A) INTRUSIVE TECHNIQUES

Backpack Electrofishing

Key References

- Bonar et al. (In Press), Johnson et al. (2007)

Advantages

- Demonstrated effectiveness over a wide range of species and habitats
- Capture efficiency can be higher than other techniques under optimal conditions
- Can be used when water clarity prohibits visual observation techniques
- Allows capture of large number of fish for further study (i.e. can get length and weight measurements, and can mark fish for mark recapture studies)

Disadvantages

- Difficult to conduct in remote areas
- Can cause significant harm to fish and eggs
- Is selective – larger fish and particular species may be more susceptible to capture; small fish, especially fry may be unaffected
- Ineffective in deep water and when water conductivity < 100 or > 500 µmhos/cm
- Ineffective in water temperatures below 4 °C
- Ineffective in steep gradient, turbulent water
- Ineffective in excessively turbid water (< 30 cm visibility##)
- Requires relatively high degree of crew training for safety and proficiency

Appropriate Situations for Use

- Most small streams < 1.5 m depth

- Water visibility $\geq$ 30 cm
- For inventory of species that are NOT listed under ESA or other state conservation legislation – consult local permitting officials regarding appropriate sample methods for listed species

Sampling Operations

- A crew should consist of a minimum two individuals; an operator and netter
- The crew should work from downstream to upstream so that disturbed debris and sediment does not interfere with catching fish as the material drifts downstream
- Downstream sweeps can be effective when stop nets are used
- Each crew member should wear polaroid glasses as this will increase the ability to see fish in the water
- The shocker can be secured upright on the stream bank (if the cable for the anode is long enough) or the operator can carry the unit on his or her back
- Walk slowly through the water, making sure both the cathode and anode are in the water
- Slowly swept the anode from side to side, with a general motion of drawing the anode towards the operator
- This motion will help to attract fish to the anode

- The power switch should be turned on and off since continued application of electrical current to the water will cause herding behavior of the fish and reduce catch efficiency
-

Fish Handling

- For presence/absence surveys record the species of each fish and other desired information
    - Keep fish in a handling bucket or live well until the sample unit is completely sampled
    - Monitor the condition of captured fish regularly as aeration may be required in warmwater conditions
    - A voucher specimen should be collected for any difficult to identify or new species for the study (check that you have the proper permits first)
- For depletion and mark-recapture surveys
    - Same as above plus: mark each species according to regional and district protocols
    - Be sure to contact other agencies and organizations to coordinate marking programs

Seines

Key References

- Little et al. (1984), Patton et al. (2000) Johnson et al. (2007)

Advantages

- Easily deployed with minimal training
- Good for less mobile, small, schooling fish

Disadvantages

- Limited number of habitat types where it is applicable
- Fish can escape at different rates depending on the obstructions present
- Difficult to keep consistent effort across habitat types
- Larger fish can avoid nets more than smaller fish
- Fish injury is common especially small/young fish as the seine is closed

Appropriate Situations for Use

- Slower moving water and back channels with few obstructions
- Silt, sand, and gravel substrates
- When target species forms schools
- In low visibility streams
- As a secondary technique when estimating species richness and distribution

Sampling Operations

- Should use seines in 10 m length increments (10, 20, 30 m, etc.); depth of seine can vary as long as the seine is as deep or deeper than the stream depth
- Mesh sizes are dependent on target species
- Start at the upstream end of the sample unit

- Pull the seine out perpendicular to the shore, keeping one end of the seine secure on shore
- While keeping the lead line on the bottom and the cork line on the surface, begin fishing the seine downstream by moving both ends downstream at a fast walking pace.
- Use 25 m fishing passes (i.e. walk downstream for 25 m) for 10 m nets and 50 m passes for larger nets
- After a 25 m fishing pass (for 10 m net) bring the offshore end of the seine downstream and back up towards the other end of the seine in a j shape to close the seine on the shore where you started, all the time keeping the lead line on the bottom and the cork line at or above the surface
- Pull the lead line carefully up onto shore forming a purse where the captured fish should collect
- If there are a lot of fish remove them from the net and place in buckets and aerate as necessary
- If there are more than one type of habitat per sample unit sample each type in the same manner
- If multiple passes of the same habitat type are to be conducted use similar pass sizes each time (i.e. for 10 m nets use 25 m passes each time)

Fish Handling

- See electrofishing methods

Plot/Quadrat Sampling

Key References

- Kessler et al. (1995), Peterson and Rabeni (2001), Weddle and Kessler (1993)

Advantages

- Technique specifically designed particularly for bottom dwelling fish
- Electric quadrats can reduce fright bias because they can be triggered remotely
- Can be more efficient than other techniques for specific species under specific conditions (i.e. large cobble substrate)

Disadvantages

- Assumes placement of quadrats does not effect fish behavior
- Relatively time consuming and requires more training and equipment
- Requires secondary technique to calibrate capture efficiency

Appropriate Situations for Use

- Abundant benthic fishes present, such as family Cottidae, Cyprinadae, Ictaluridae, and Percidae
- Large boulder and cobble riffle habitat
- Population estimates by area are required

Sampling Operations

- Construct 1 m^2 quadrats using PVC pipe as per Peterson and Rabeni (2001)
- Use 0.75-m-deep collection bag attached to the back of the sampler for collecting fish
- Place the sampler in a riffle, securing it to the streambed

- Collected fish by trapping them within the sampler and driving them into the collection bag
- Disturb the substrate within the sampler by kicking to dislodge fish and moved them into the collection bag
- Start at the downstream end of a riffle
- Pace individual quadrat subsamples at uniform intervals longitudinally and laterally to ensure good coverage of the stream

Fish Handling

- See electrofishing methods

B) NON-INTRUSIVE TECHNIQUES

Underwater Observation (Snorkel Surveys)

Key References

- Throw (1994), Dolloff et al. (1996)

Advantages

- Easily adapted to a variety of applications
- Can estimate behavior, habitat use, fish size structure, and gear performance
- Non-intrusiveness and less destructive
- Modest personnel and gear requirements
- Reduced costs

- Appropriate when extreme conductivity (low or high), habitat complexity, or depth limit other techniques

Disadvantages

- Need clear water conditions
- Water temperatures should exceed 4-9 °C depending on fish species to optimize fish detection
- Does not allow fish capture, therefore may be unable to accurately determine size, sex, age, reproductive status, or identify small individuals or cryptic species
- Can pose serious hazards if basic safety considerations are not properly addressed

Appropriate Situations for Use

- Water clarity must be sufficient to enable observers to see the stream bottom in the deepest sampling units
- When species can accurately be identified to species by observation only
- When avoidance of the observer by fish can be detected
- Visibility of 2-3 m usually meets these criteria

Sampling Operations

- Observers should routinely measure and record the visibility of a known object prior to sampling
- A suitable object is a fish silhouette with distinguishing markings

- Estimate visibility with a secchi-disk like approach that averages three measurements of the maximum distance at which the marks on the silhouette are visible
- Area to be sampled must have sufficient depth to enable the observers to submerge a mask
- Optimum day-time sampling light conditions 1000-1700 hours
- Observations conducted at night or during twilight hours require hand-held or fixed position underwater lights
- Divers typically enter the water downstream from the area to be sampled
- After entering the stream, the observer pauses to acclimate and to allow any organisms disturbed by the initial approach to resume normal behavior
- Move slowly upstream and avoid sudden movements
- When it is impractical to move upstream as a result of current or depth, observers may float downstream with the current, remaining as motionless as possible
- A zigzag pattern is often used between banks, taking care to thoroughly search stream margins and all cover such as undercut banks, substrate interstices, and woody debris
- Record target fish species and size class (predetermined)
- Survey areas of good visibility before moving into areas with lower visibility
- Shallow water habitats such as riffles typically require more observers than deeper habitats

- When using more than one diver divide the stream into lanes with each diver counting all fish within an assigned lane
- If the unit is too turbulent or complex, natural features such as a line of boulders can be used to partition the unit
- The distance between observers should always be no greater than the maximum underwater visibility
- Observers must start and stop at the same time
- Remain in their assigned lanes, and move at the same speed
- Observers must be careful to avoid counting fish that move among lanes to avoid counting the same fish twice.

Nest or Redd Counts

Key Reference(s)

- Bonar et al. (In Press), Johnson et al. (2007)

Advantages

- Non-intrusiveness and less destructive
- No gear requirements
- Limited observer training required
- Reduced costs

Disadvantages

- Redds may not reliably indicate population status because:
  - One female can produce several discrete redds or a single large structure

- not all fish construct redds and that there is great variation in how redds may be constructed within species
- Sometimes females will construct “test” digs in which eggs are not deposited
- Can be difficult to accurately count the number of redds where they are clustered close together or superimposed
- The number of redds that individual females construct can vary
- Stream hydraulics, disturbance by other animals may cause sites to look like fish redds

Appropriate Situations for Use

- Water clarity must be sufficient to enable observers to see the stream bottom in the deepest sampling units
- Visibility of 2-3 m usually meets these criteria
- Most likely sites include pool tailouts, riffles, lower gradient channels, tributary and debris flow confluences, and sites adjacent to instream objects, such as wood and boulders

Sampling Operations

- Recommend pre-spawning surveys to familiarize observer with stream habitat
- Criteria for recognizing redds:
  - Presence of adult salmonids near a suspected redd
  - Disturbed gravel exhibiting an elliptical shape and oriented directly into the current

- 3-dimensional morphology with the pit and tailspill clearly visible (Figure 10)
- Appropriate gravel size for the target species
- Size of the redd is appropriate for the target species

- Conduct counts on the ground from stream banks or by carefully wading
- Following the pre-spawning survey, conduct multiple passes through sample sites over the duration of spawning and during each pass:
    - Mark locations of redds on the site map or sketch
    - Flag redd locations with unique identification codes marked on the flag with a waterproof marker
    - The number of survey passes depends on the extent of the spawning period and the detection life or duration of redd visibility
    - More passes may be necessary if the spawning season is protracted or if conditions affecting detectability of redds change
    - Multiple counts have the advantage of accounting for variation in spawning timing and allowing observers to track the development of redds and more accurately count completed redds
- Redd counts can be conducted by walking in an upstream or downstream direction
- Avoid disturbance of stream substrates that could obscure visibility and disturb fish
- When multiple species construct redds in similar locations and at similar times conduct frequent (e.g., 4-8d) surveys to closely track redd

accumulations and to increase the chances that adults of a given species can be identified on redds

- Also frequent redd counts may be needed in locations where there may be a high probability of redd superimposition due to large numbers of spawners or very limited availability of spawning habitat
- Optimal timing of sampling depends on the species in question

Figure 10. A. Typical salmonid redd including the excavated pit and egg pocket (to be created).

Minnow Traps

Key Reference(s)

- Bryant (2000)

Advantages

- Easy to deploy
- Limited training required

Disadvantages

- Limited to smaller fish species or age classes
- Limited habitat types can be sampled (usually slower moving streams)
- Not logistically feasible to sample remote areas efficiently
- Need to minimum of 12 hours to fish effectively

Appropriate Situations for Use

- Slower moving streams

- Abundant small fish or younger age classes present
- Time and access not restricted
- Deep water or complex habitat (abundant undercut banks or LWD) where fish can escape other techniques

Sampling Operations

- Position trap on the bottom or suspended at a particular depth
- Traps can be set with or without bait
- Conlin and Tutty (1979) suggest that the most effective use of a minnow trap includes using fresh roe as bait and soak time of 24 hours
- Other baits include preserved fish roe, catfood, sardines, canned fish, corn, shrimp and cheese
- Traps are set in a variety of habitats such as weeds, beach areas, under overhanging vegetation or amongst submerged logs
- All traps **must** be recovered as they continue to fish indefinitely
- Small rocks can be put into the trap to provide some refuge for the smaller fish

Fish Handling

- See electrofishing methods

Appendix 6. Field data forms for distribution, abundance, and trend surveys.

To be completed.

Appendix 7. Computer programs for data analysis.

Software Name (Author)	Intended Use	Web Address
MICROFISH (VanDeventer and Platts 1989)	Population estimates from depletion data using maximum likelihood estimation	http://microfish.org/
CAPTURE, MARK (Otis et al. 1978, White and Burnham 1997, Cooch and White 2006)	Population estimates and many other population parameters, including survival, from mark recapture data	http://www.warnercnr.colostate.edu/~gwhite/software.html
DISTANCE (Laake et al. 1994)	Population estimates from line transect and distance data	http://www.ruwpa.st-and.ac.uk/distance/
MONITOR (Gibbs and de Arellano 2007)	Estimates the statistical power of monitoring programs	http://www.esf.edu/efb/gibbs/monitor/
SPECIES DIVERSITY (Krebs 1999)	A variety of applications based on the book Ecological Methodology	http://exetersoftware.com/cat/ecometh/ecomethodology.html
PRESENCE (Mackenzie et al. 2002)	estimates the probability a site is occupied by a species (and related parameters), given that the species will not always be detected with certainty, even when present	http://www.proteus.co.nz/home.html
TRENDS (Gerrodette 1993)	implements power analysis for detecting trends in abundance using linear regression	http://swfsc.noaa.gov/textblock.aspx?Division=PRD&ParentMenuId=228&id=4740
Various	Calculating statistical power; review by Thomas and Krebs 1997	http://www.zoology.ubc.ca/~krebs/power.html

Appendix 9. Formulas for calculating confidence limits.

To be completed.

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