Body condition indices as a rapid assessment of the abundance of introduced salmonids in oligotrophic lakes of southern Chile

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Abstract

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In South America, salmonids have been widely introduced to lakes, yet there are still many oligotrophic lakes without current abundance estimates. Catch per unit effort is a good estimator of fish abundance, but its correct implementation requires an expensive and long-term effort. Thus, there is a need to develop a rapid assessment to estimate the relative abundance of salmonids. Under well-delimited circumstances, condition indices may be a good technique to estimate fish population relative abundances. We sampled adult salmonids with gillnets from 6 oligotrophic lakes in southern Chile during the Austral summer. Our data show that as rainbow trout condition indices decrease, the abundance of salmonids increases both temporally and spatially. Because all 6 lakes are limited by food, the most likely reason for the change in the rainbow trout condition indices is change in salmonid abundance. The strong inverse relationship between condition indices of rainbow trout and the relative abundance of salmonids suggests that our model can be a useful tool for the estimation of relative fish abundance in lakes. This study provides a model that can be used to improve lake and fishery management decisions for introduced salmonids in southern South America.

Key words: gillnet, invasive species, Patagonian lakes, rainbow trout, recreational fisheries, salmon aquaculture

Relative abundance, expressed as catch per unit effort (CPUE), is often used as an index of population density. Its correct implementation requires a long-term continuous and systematic effort that implies a great expense (Murphy and Willis 1996, Guy and Brown 2007). In South America, the use of CPUE is limited because of a lack of long-term management strategies and the relatively high cost of their design and implementation. However, there is a current need to better estimate the relative abundance of introduced game species and thus improve the management of several emerging inland recreational fisheries in this part of the world

(Vigliano et al. 2000, Shrestha et al. 2002, Arismendi and Nahuelhual 2007, Núñez and Niklitschek 2010).

In lakes of the Patagonia, salmonids (salmon and trout) have been extensively introduced beginning in the early 1900s, resulting in rainbow trout (*Oncorhynchus mykiss*) becoming (1) the most common fish (Soto et al. 2006, Aigo et al. 2008); (2) self-sustaining (Soto et al. 2006); and (3) the most desired fish to catch (Arismendi and Nahuelhual 2007). In Chile, further introductions of rainbow trout and other salmonid species (Basulto 2003) began with modern salmon aquaculture in the 1980s. Salmonid species are escaping from aquaculture net-pens and supplement the fish stock available to anglers. These escaped fish include Coho salmon (*O. kisutch*), Chinook salmon (*O. tshawytscha*) and

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Atlantic salmon (*Salmo salar*; Soto et al. 2006, Arismendi et al. 2009). Abundance levels in some lakes may be at their carrying capacity limit (Arismendi et al. 2009).

At present, the salmonid recreational fishery represents an important economic activity in southern South America (Vigliano et al. 2000, Arismendi and Nahuelhual 2007, Núñez and Niklitschek 2010), but many oligotrophic lakes lack published information on the current abundances of introduced salmonids (Pascual et al. 2002, Soto et al. 2006, Aigo et al. 2008, Arismendi et al. 2009). Specifically in Chile, at least 9 of the 15 lakes >10 km^2 in surface area located 39–43°S lack fish abundance data. Previous studies also have resulted in data that are difficult to compare because of varying use and application of capture techniques (Quirós 1998, Soto et al. 2006, Baigún et al. 2007). Thus, there is a pressing need to rapidly estimate the relative abundance of salmonid populations to effectively manage those fish populations in southern South America.

Condition indices have not previously been used to estimate relative population abundances, but they may prove to be a useful tool to estimate relative fish abundance under well-delimited conditions. Because condition indices depend mainly on both the relative abundance of food and the number of individuals using the resource (Bennett 1970), the well-being of the fish can be directly attributed to their population abundance. Condition indices provide information about the fish body, with a good condition indicating favorable food availability or a sparse abundance of fishes, and a poor condition indicating an overpopulation of fishes (Bennett 1970, Blackwell et al. 2000).

Two body condition indices are widely used to measure the well-being of fishes (Blackwell et al. 2000, Froese 2006): Fulton's condition factor or K factor (Ricker 1975), and relative weight or Wr (Wege and Anderson 1978). The K factor is the ratio between a fish's weight and length raised to the third power and works under the assumptions of isometric growth that heavier fishes of a given length are in better condition. The Wr index is the ratio between the actual fish weight and a standard weight for a fish of the same length.

In general, "condition indices" have created ongoing confusion about their correct application and interpretation (Froese 2006) because they may depend on several variables. For instance, K factor may vary by season (Trudel et al. 2005), animal health status (e.g., Neff and Cargnelli 2004), differences in growth rates between juvenile and adults (He et al. 2008, Rennie and Verdon 2008), life-history changes in the animal body form (e.g., during the reproductive season; Blackwell et al. 2000) and changes in food availability (Pothoven et al. 2001, Rennie and Verdon 2008). However,

condition indices can be useful if used correctly by minimizing sources of variability. For example, the K factor has been used as an indicator of recent feeding activity (Cone 1989), body energy reserves (Neff and Cargnelli 2004, Pangle and Sutton 2005, Herbinger and Friars 2008), energy densities (Trudel et al. 2005) and physiological conditions (Rennie and Verdon 2008). In comparison, the advantages of Wr are that the standard weight compensates for inherent changes in the body form, and its values can be compared between fish of different lengths and from different populations (Ney 1999, Blackwell et al. 2000). Recently, condition indices have also been used to track recent ecological declines in the well-being and growth of lake whitefish (*Coregonus clupeaformis*) populations in the Great Lakes (Pothoven et al. 2001, Lumb et al. 2007, DeBruyne et al. 2008).

In this study, we measure the well-being of adult rainbow trout in temperate oligotrophic lakes of southern Chile using 2 body condition indices: K factor and Wr. Because historically the productivity of these lakes has been described as limited (Thomasson 1963, Campos 1984, Soto 2002, Steinhart et al. 2002, De Los Rios and Soto 2007), we expect that changes in K factor and Wr are a result of changes in relative salmonid abundance rather than changes in food availability. We hypothesize that both K and Wr indices of rainbow trout will display a strong and inverse relationship with relative salmonid abundance. Thus, if our hypothesis is true, condition indices could be used as a rapid estimation of relative salmonid abundance in temperate oligotrophic lakes of southern South America. Fishery managers of these oligotrophic temperate lakes in southern Chile sought to identify (1) the relative abundance of introduced salmonids in lakes where there is a lack of biological information, (2) the potential for competition among salmonids species, and (3) how net-pen escapees may influence the quality of recreational fishery of rainbow trout.

Study sites

We sampled fishes from 6 lakes in the Araucanian Patagonian Lake District (Thomasson 1963, Campos 1984, Geller 1992) of southern Chile, South America (39–43°S; Fig. 1) because these lakes have the largest surface area of all lakes in the region and are most frequented for sport fishing in the region. In this area, the climate is temperate with an annual range of precipitation of 1300–2200 mm. All Araucanian Patagonian lakes have common characteristics, including that they are (a) of glacial origin with volcanic influence in the soils; (b) deep ($Z_{max} >100$ m); (c) warm monomictic with thermal stratification in the summer; (d) high in water transparency; (e) low in salt and nutrient concentrations; and
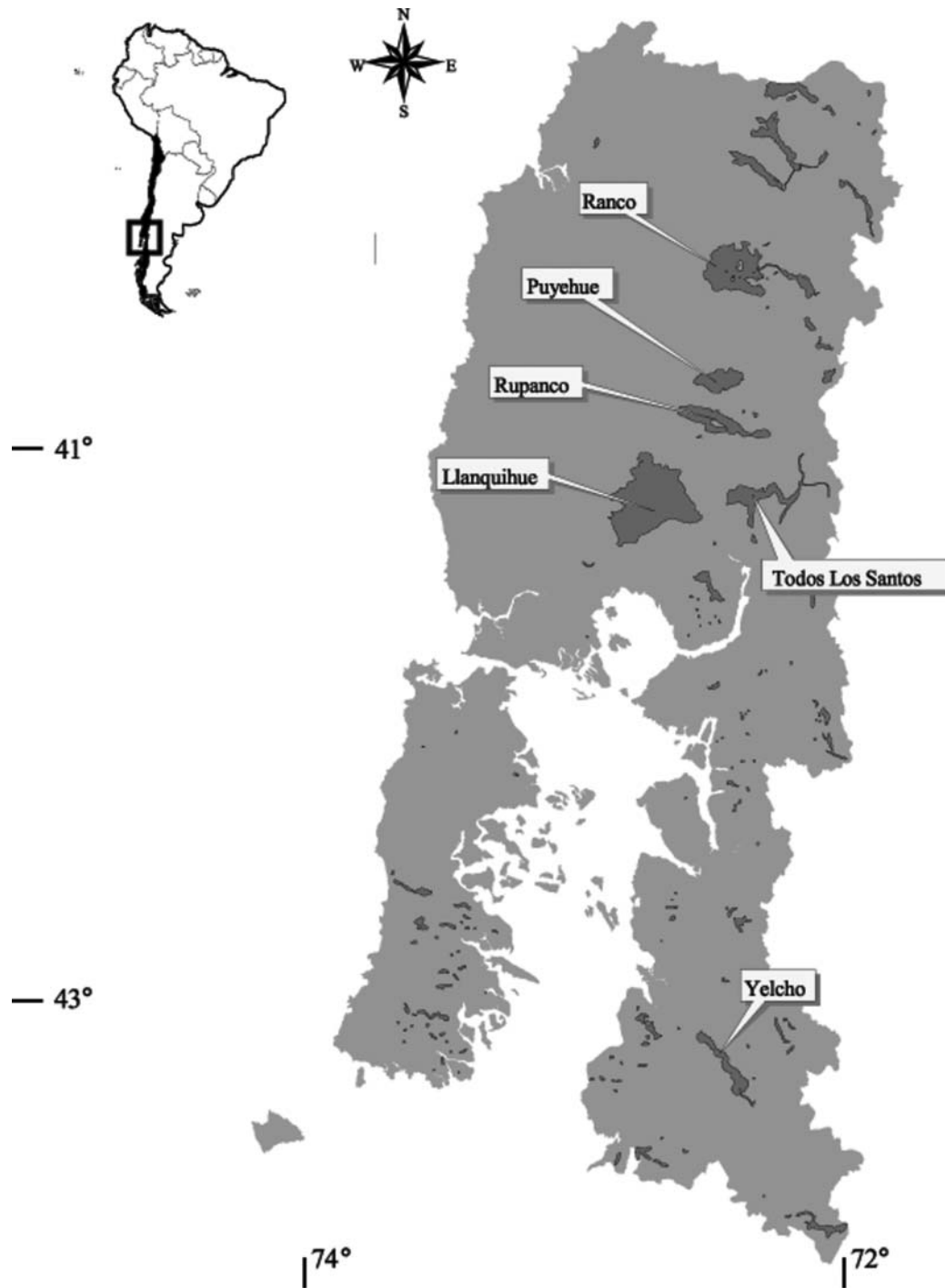
Figure 1.-Location of six lakes sampled in the Lakes District, southern Chile.

(f) oligotrophic with very low chlorophyll-a concentrations (more limnology details are provided by Thomasson 1963, Campos 1984, Geller 1992, Woelfl 2007). The 6 lakes are located at 44–189 m a.s.l., with surface areas ranging from 119 to 870 km² and mean depths between 76 and 191 m.

Materials and methods

Fish capture and condition indices

We conducted 4 fishing expeditions in each lake from 2000 through 2001. Previous samples for Lake Llanquihue were included, adding an additional 19 fishing expeditions from

1993 through 1999. For all fishing expeditions, we used two 280 m long × 3.5 m deep experimental gillnets of varying stretched mesh sizes (3.81, 7.62, 12.06 and 15.24 cm), positioned perpendicular to the shoreline of the lake beginning at the water's edge and checked every 4 h for 48 h. Captured salmonids were identified to species, weighed (g) and measured (cm; total length). We defined CPUE as the number of fish per fishing hour and used it as an indirect measure of the population abundance. The sampled lakes were large (surface area >119 km^2), and the fishing effort was a relatively short-time period; therefore, we did not consider that fish populations were affected by our fishing method.

We minimized the potential variability of other factors associated with fish condition indices so that rainbow trout condition indices mainly depended on changes of salmonid abundance. To reduce the influences of season (e.g. Trudel et al. 2005) and the life-history changes in body form (e.g. Blackwell et al. 2000), we used the information obtained from salmonids captured only during the nonreproductive summer season (Dec–Mar). Further, to avoid the influence of differences in growth rates between juvenile and adults (e.g. He et al. 2008, Rennie and Verdon 2008), we only considered adult salmonids (>30 cm total length, the minimum take size allowed by the Chilean law). In addition, because previous studies conducted in our study lakes reported no evidence of potential diseases or health problems in salmonid populations (Soto et al. 2006, Arismendi et al. 2009), we assumed no influence of fish health in the estimations of condition indices for this study (e.g. Neff and Cargnelli 2004). Finally, because the lakes in southern Chile are oligotrophic and have very low productivity (Thomasson 1963, Campos 1984, Soto 2002, Steinhart et al. 2002, De Los Rios and Soto 2007), the salmonid diet is based on aquatic insects, freshwater crabs and small-sized native fishes (Arenas 1978, Campos 1984, Arismendi et al. 2009), and most likely diets do not have the ability to change abruptly because the available food resources are so limited. Therefore, in our study we assumed that food availability has a low potential to influence rainbow trout condition indices (e.g., Pothoven et al. 2001, Rennie and Verdon 2008).

We determined the condition factor (K) for each fish captured (>30 cm total length) as

$$K = \left(\frac{W}{L^3} \right) \times 100 \qquad (1)$$

where W represented actual fish weight (g) and L total length (cm). The K factor was multiplied by 100 to get values near unity. Also, for each rainbow trout individual (>30 cm total length), the relative weight (Wr) was determined with the
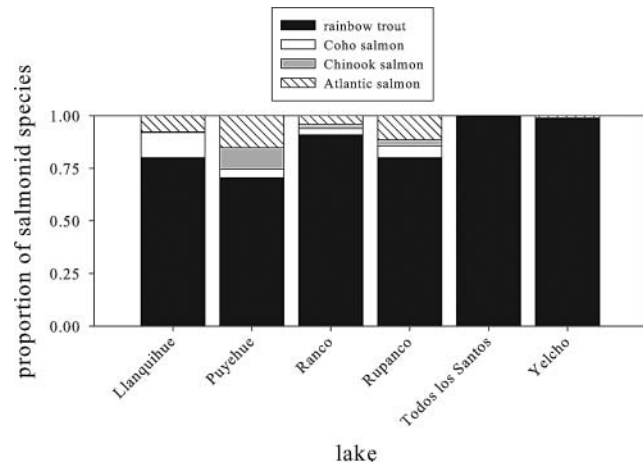
Figure 2.-Proportion of salmonid species found in the six lakes sampled from the Lakes District, southern Chile.

following formula:

$$W_r = \left(\frac{W}{W_s} \right) \times 100 \qquad (2)$$

where W was the actual fish weight, and Ws was the fish standard weight for a fish of the same length. The Ws estimation was based on the equation provided by Simpkins and Hubert (1996) for rainbow trout in lentic ecosystems.

Analysis

To show that the rainbow trout length comparison among lakes is valid (He et al. 2008, Rennie and Verdon 2008), we tested potential differences in the rainbow trout length data using the nonparametric Kruskal-Wallis median test. For Lake Llanquihue, we performed a simple regression analysis between both condition indices and the total length of rainbow trout. If no association was found between either condition index and the total length of rainbow trout, then we could assume that any change in a condition index was not influenced by changes in the length distribution of rainbow trout (e.g. to ensure that the increase in smaller fish that must be recruiting to the population was not the cause of the decrease in K and Wr).

To test our hypothesis, we performed a least squares simple linear regression analysis with K factor and Wr of rainbow trout as independent variables and CPUE of salmonids as the dependent variable. We used the condition indices of rainbow trout as the independent variable in the model because this species inhabits most of the Araucanian lakes (Soto et al. 2006, Arismendi et al. 2009) and represented more
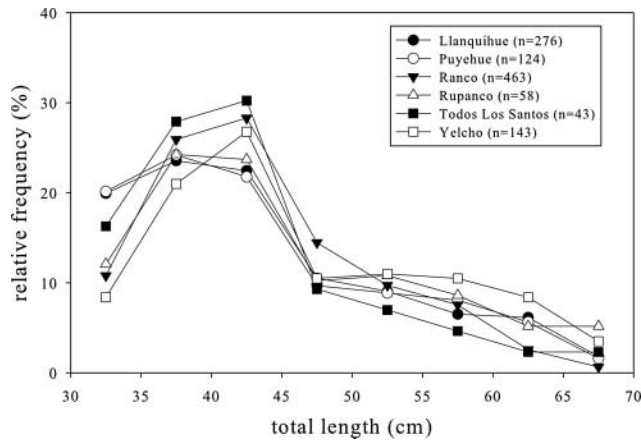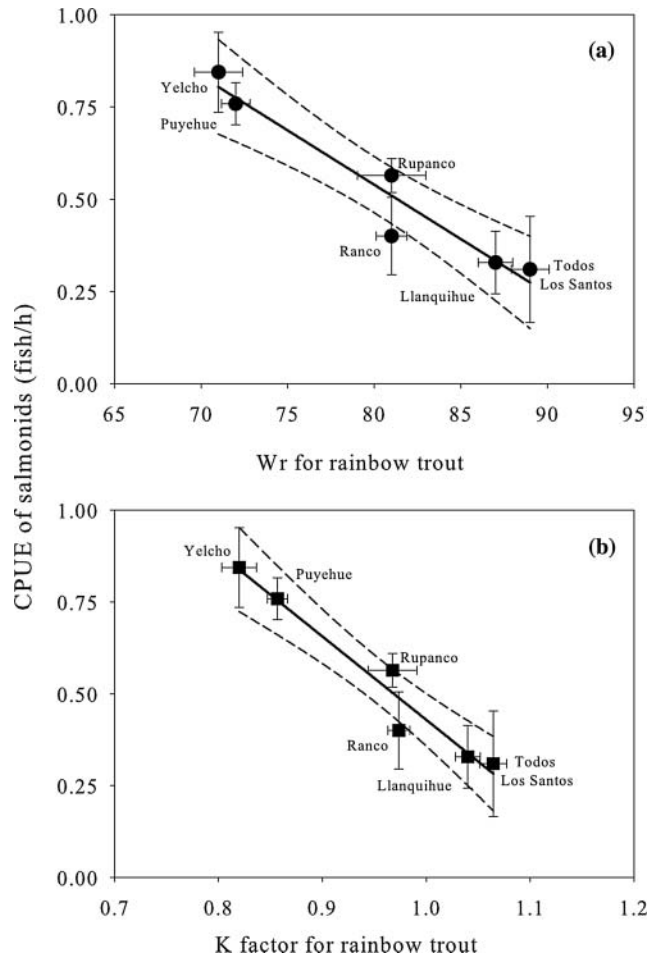
Figure 3.-Rainbow trout relative length frequency (>30 cm) for the six lakes sampled in the Lakes District, southern Chile. Each point marks the center of the length class grouped every 5 cm.

than 70% of the salmonids caught in our study (Fig. 2). We used CPUE of all salmonids (rainbow trout, Coho salmon, Chinook salmon and Atlantic salmon) as the dependent variable representing the abundance of all fish game species. We have not presented Wr values for species other than rainbow trout because there is no available information for the estimation of their specific standard weight (Blackwell et al. 2000); further, these other salmonids species had a low CPUE.

Results

Rainbow trout had a similar length distribution among lakes, with the highest proportion of individuals from 35 to 45 cm (Fig. 3; $P = 0.21$). On a spatial scale among all 6 lakes, we found strong evidence of a negative association between the mean of both K factor and Wr of rainbow trout and CPUE of salmonid species (Fig. 4). Specifically, a one-level increase in the rainbow trout K factor was associated with a decrease in mean CPUE of salmonids of 2.27 units; and a one-level increase in the rainbow trout Wr was associated with a decrease in mean CPUE of salmonids of 0.029 units. Overall, the mean CPUE of salmonid species is explained by 94% of the variation in the rainbow trout K factor and 91% of the variation in the rainbow trout Wr.

Rainbow trout (Fig. 5) and the other salmonid species (Coho salmon, Chinook salmon and Atlantic salmon) had similar K factor values ranging between 0.5 and 2.0. At Lake Llanquihue, Chinook salmon had the higher K factor values (mean 1.18 ± 0.22 SE), followed by Coho salmon (mean 0.98 ± 0.01 SE) and then by Atlantic salmon (mean 0.88 ± 0.03 SE). At Lake Puyehue, we found higher K factor values for Coho salmon (mean 1.18 ± 0.02 SE), followed by Chinook

Figure 4.-(a) Scatterplot of mean CPUE of salmonids (fish/h) and mean Wr of rainbow trout with a 95% confidence interval band for estimated means (dotted lines) in six lakes of southern Chile. Simple linear regression model:
$CPUE\_salmonids = 2.89 - 0.029 \times W_r\_rainbow\_trout$ ($R^2 = 0.93$; $R^2_{ad} = 0.91$; $P = 0.002$). (b) Scatterplot of mean CPUE of salmonids (fish/h) and mean of K factor for rainbow trout with a 95% confidence interval band for estimated means (dotted lines) in six lakes of southern Chile. Simple linear regression model:
$CPUE\_salmonids = 2.70 - 2.27 \times K_{f\,actor}\_rainbow\_trout$ ($R^2 = 0.95$; $R^2_{ad} = 0.94$; $P = 0.0009$).

salmon (mean 1.11 ± 0.01 SE) and then Atlantic salmon (mean 0.92 ± 0.03 SE). At Lake Rupanco, Coho salmon had the highest k factor values (mean 1.29 ± 0.11 SE), followed by Atlantic salmon (mean 1.09 ± 0.03 SE) and then Chinook salmon (mean 0.99 ± 0.01 SE). At Lake Ranco, the K factor values were highest for Atlantic salmon (mean 1.15 ± 0.05 SE), followed by Chinook salmon (mean 1.03 ± 0.04 SE) and then by Coho salmon (mean 0.97 ± 0.05 SE). At Lake Yelcho, Atlantic salmon had a K factor of 0.96 ± 0.07 SE, and at Lake Todos Los Santos we did not find any other salmonid species.
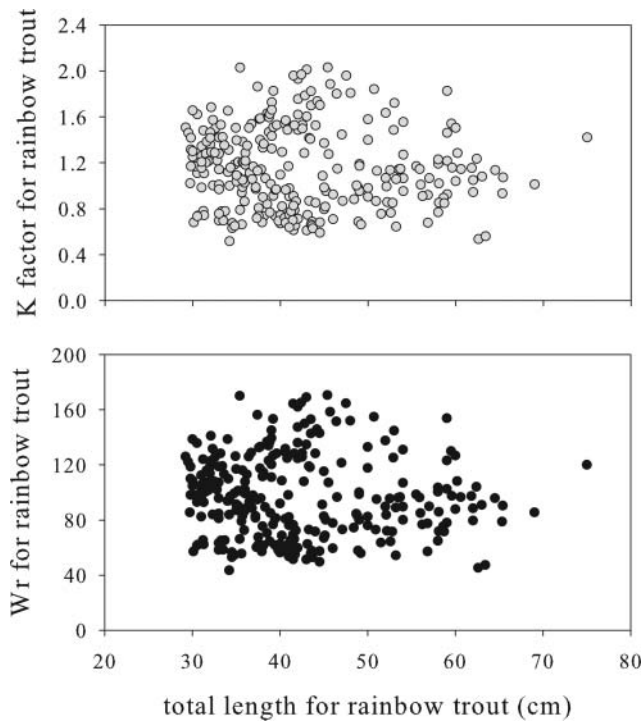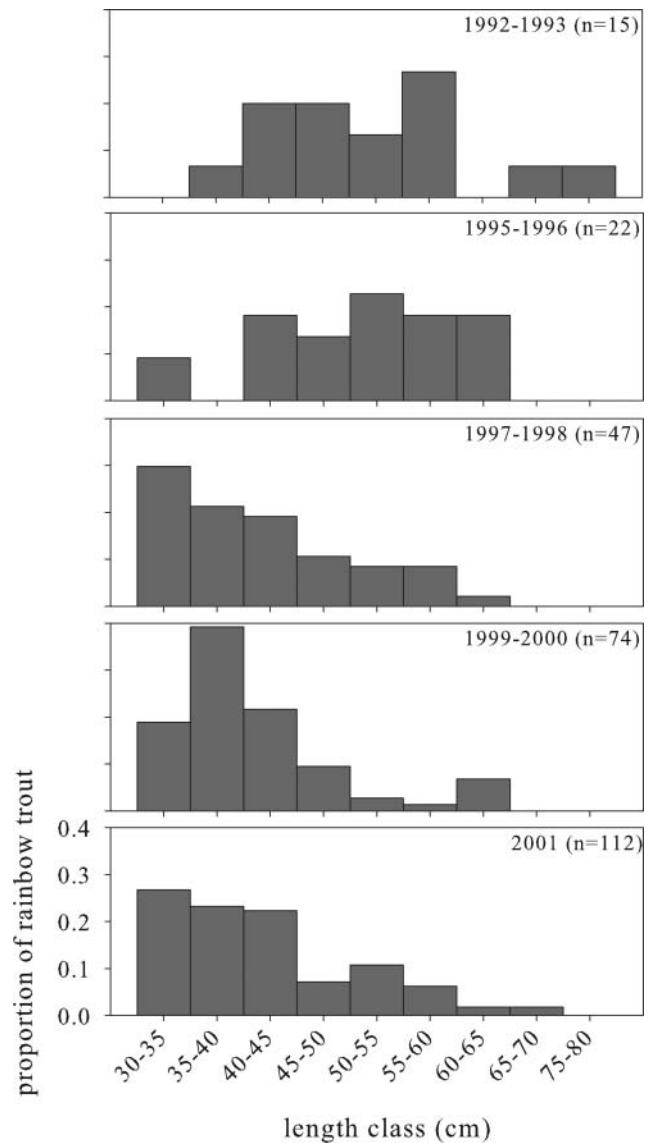
Figure 5.-K factor and Wr condition indices of rainbow trout by length (cm) in Lake Llanquihue.

For rainbow trout in Lake Llanquihue (Fig. 5), we did not find an association between the total length and its K factor ($R^2 = 0.005$; $R^2_{adj} = 0.001$; $P = 0.23$ or its Wr condition index ($R^2 = 0.002$; $R^2_{adj} = 0.001$; $P = 0.37$). We also found an increase in the capture of small trout over time (principally 30–45 cm TL; Fig. 6). But, similar to our spatial scale, a temporal scale across years (1993–2001) showed a decreasing trend in K factor and Wr of rainbow trout, whereas the relative abundance of salmonids (CPUE) increased (Fig. 7).

Figure 6.-Proportion of captured rainbow trout by length class (grouped every 5 cm) over time in Lake Llanquihue.

Discussion

The strong inverse relationship between condition indices of rainbow trout and salmonid relative abundance (CPUE) suggests that our model can be a useful tool as a rapid estimation of relative fish abundance for management of recreational fisheries. Our approach is valid as an index of population density and has yet to be validated to estimate absolute population sizes. Our data show that as rainbow trout condition indices decrease, the relative abundance of salmonids, expressed as CPUE, increases both temporally, as is the case of Lake Llanquihue, and spatially, considering the 6 sampled lakes. Because all 6 of these oligotrophic lakes are limited by food (Thomasson 1963, Campos 1984, Soto

2002, Steinhart et al. 2002, De Los Rios and Soto 2007), the most likely reason for the change in the rainbow trout condition indices is change in salmonid abundance.

The temporal patterns of salmonid abundance and rainbow trout condition in Lake Llanquihue support the idea that a decline in the condition of fishes in these lakes is coupled with an increase in relative abundance. This suggests the occurrence of density-dependent mechanisms such as those proposed in the case of *C. clupeaformis* in Lake Michigan in North America (DeBruyne et al. 2008). Density-dependent reductions in rainbow trout condition indices can result from competition for food, which is thought to be limiting in our
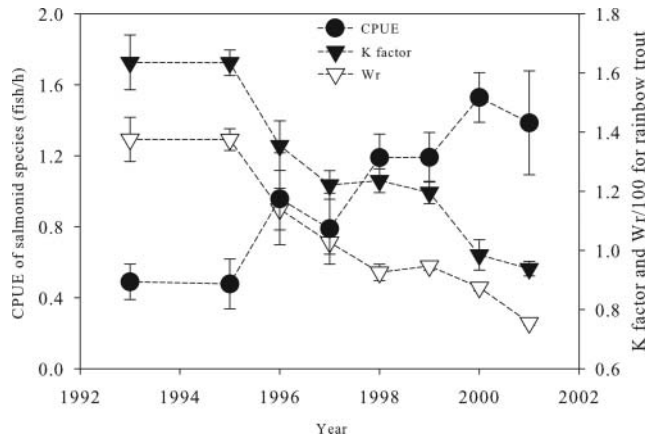
Figure 7.-Yearly distribution of mean CPUE of salmonids (±SE; fish/h) and mean of condition indices Wr (±SE) and K factor (±SE) for rainbow trout (>30 cm) in Lake Llanquihue, southern Chile between 1993 and 2001.

study sites. Many studies have suggested that food resources may limit salmonid abundances in lakes (Lawrie and Rahrer 1972, Fausch and White 1986, Scott et al. 2003).

Although we show that rainbow trout condition is dependent on salmonid abundance, the presence of more salmonids is not the only factor that can limit fish condition. Other factors that we did not consider can influence that relationship, such as the season, animal health status, differential growth rates between juvenile and adult fishes, life-history changes of animal body form and food availability (Blackwell et al. 2000, Pothoven et al. 2001, Neff and Cargnelli 2004, Trudel et al. 2005, He et al. 2008, Rennie and Verdon 2008). We could not control for animal health, but we controlled for season, fish size, fish growth and animal body forms by limiting the scope of our study to the abundance of adult salmonids during the Austral summer. Our results in the range of lengths used here show no differences in the size of salmonids among lakes. In Lake Llanquihue, trout size does not influence any condition indices temporally. Further, because our study took place in oligotrophic lakes shown in previous studies,to have consistently low productivity, we made a safe assumption that there was no change in food availability. Our approach may simplify the study systems, but our model has high predictive power showing that the rainbow trout condition and relative abundance, expressed as CPUE, are inextricably linked, and therefore one can be used to predict the other.

Salmonid abundance levels estimated from our model could be used by managers to influence stocking decisions and fishing pressure. Using our model with potential creel data from anglers' trout catches (i.e., species identification, weight and length sizes), fisheries managers could calculate rainbow trout condition and estimate the relative abundance

of salmonids. Then, depending on lake-specific management plans, managers would modify or adjust the fishing regulations for the next fishing season (e.g., change the total fishing quota, extend or restrict the fishing season period, and/or change the size of trout that anglers are allowed to keep). Increasing the individual fish size and the number and well-being of fish caught may also result in economic benefits (Lawrence 2005, Loomis 2006). Thus, by establishing the specific goals for the next fishing season (e.g., increase in trout relative abundance or improvements in trout condition indices) and using additional lake-specific biological information (e.g., species composition and length distributions), fisheries managers can use an adaptive management based approach to integrated lake management over time.

Specifically, our model can be used as part of an innovative management approach for recreational fisheries and natural resource conservation in oligotrophic lakes in southern South America to estimate the abundance of escaped salmonids in a lake, especially as more lakes are being considered for in-lake aquaculture production (León-Muñóz et al. 2007, Arismendi et al. 2009). Salmonid invasions have been extremely successful in southern South America (Pascual et al. 2002, Soto et al. 2006, Arismendi et al. 2009), and fisheries managers are in dire need of an effective management approach to manage recreational fisheries, especially because local and regional economies depend on these recreational fisheries (Vigliano et al. 2000, Arismendi and Nahuelhual 2007, Nuñez and Niklitschek 2010).

In Lake Llanquihue, for example, if the management goal is to improve the future condition of rainbow trout, then fishery managers could promote an increase in the capture of salmon species by extending the fishing season and increasing the fish quota, especially Coho and Atlantic salmon, which have not been found to have reproducing populations in Southern Chile (Soto et al. 2006, Arismendi et al. 2009). Our study showed an increase in salmonid relative abundances and a decrease in the condition of rainbow trout over time. As a consequence, anglers have started to lose interest in Lake Llanquihue due to the current poor condition of the rainbow trout (I. Arismendi, pers. observ.); thus, our model could help set the harvest levels for fishery management plans and help control aquaculture salmonid escapees (see recommendations provided by Belle and Nash 2009). This study provides insight into the management of lake systems based on the importance of condition indices and the relative abundance of salmonids in southern South America.

Acknowledgments

References

Aigo J, Cussac V, Peris S, Ortubay S, Gómez S, López H, Gross M, Barriga J, Battini M. 2008. Distribution of introduced and native fish in Patagonia (Argentina): patterns and changes in fish assemblages. Rev Fish Biol Fisher. 18(4):387–408.

Arenas J. 1978. Análisis de la alimentación de *Salmo gairdnieri* Richardson en el lago Riñihue y río San Pedro, Chile. Medio Ambiente. 3:50–58.

Arismendi I, Nahuelhual L. 2007. Non-native salmon and trout recreational fishing in Lake Llanquihue, southern Chile: Economic benefits and management implications. Rev Fish Sci. 15:311–325.

Arismendi I, Soto D, Penaluna B, Jara C, Leal C, León-Muñoz J. 2009. Aquaculture, non-native salmonid invasions, and associated declines of native fishes in lakes of the northern Chilean Patagonia. Freshwater Biol. 54:1135–1147.

Baigún C, Oldani N, Madirolas A, Alvarez G. 2007. Assessment of fish yield in Patagonian Lakes (Argentina): Development and application of empirical models. Trans Am Fish Soc. 136:846–857.

Basulto S. 2003. El Largo viaje de los salmones: una crónica olvidada, Propagación y cultivo de especies acuáticas en Chile. Santiago (Chile): Editorial Maval Ltda.

Belle SM, Nash CE. 2009. Better management practices for net-pen aquaculture. In: Tucker CS, Hargreaves JA, editors. Environmental mest Management practices for aquaculture. John Wiley & Sons. p. 261–330.

Bennett GW. 1970. Management of lakes and ponds. 2nd ed. New York (NY): Van Nostrand Reinhold Co.

Blackwell BG, Brown ML, Willis DW. 2000. Relative weight (Wr) status and current use in fisheries assessment and management. Rev Fish Sci. 8:1–44.

Campos H. 1984. Limnological study of Araucanian lakes (Chile). Verh Internat Verein Limnol. 22:1319–1327.

Cone RS. 1989. The need to reconsider the use of condition indices in fishery science. Trans Am Fish Soc. 118:510–514.

DeBruyne RL, Galarowicz TL, Claramunt RM, Clapp DF. 2008. Lake whitefish relative abundance, length-at-age, and condition in Lake Michigan as indicated by fishery-independent surveys. J Great Lakes Res. 34:235–244.

De Los Ríos P, Soto D. 2007. Crustacean (Copepoda and Cladocera) zooplankton richness in Chilean Patagonian lakes. Crustaceana. 80: 285–296.

Fausch KD, White RJ. 1986. Competition among juveniles of Coho salmon, brook trout, and brown trout in a laboratory stream, and implications for Great Lakes tributaries. Trans Am Fish Soc. 115:363–381.

Froese R. 2006. Cube law, condition factor and weight–length relationships: history, meta-analysis and recommendations. J Appl Ichthyol. 22:241–253.

Geller W. 1992. The temperature stratification and related characteristics of Chilean lakes in midsummer. Aquat Sci. 54(1):37–57.

Guy CS, Brown ML, editors. 2007. Analysis and interpretation of freshwater fisheries data. Bethesda (MD): American Fisheries Society.

He JX, Bence JR, Johnson JE, Clapp DF, Ebener MP. 2008. Modeling variation in mass-length relations and condition indices of lake trout and Chinook salmon in Lake Huron: a hierarchical Bayesian approach. Trans Am Fish Soc. 137(3):801–817.

Herbinger CM, Friars CW. 2008. Correlation between condition factor and total lipid content in Atlantic salmon, *Salmo salar* L., parr. Aquac Res. 22:527–529.

Lawrence K. 2005. Assessing the value of recreational sea angling in South West England. Fisheries Manag Ecol. 12:369–375.

Lawrie AH, Rahrer JF. 1972. Lake Superior: effects of exploitation and introductions on the salmonid community. B Fish Res Board Can. 29:765–776.

León-Muñóz J, Teckin D, Farias A, Díaz S. 2007. Salmon farming in the lakes of Southern Chile – Valdivian Ecoregion: History, tendencies and environmental impacts. Consultancy Technical Report WWF (World Wildlife Fund), Valdivia, Chile.

Loomis J. 2006. Use of survey data to estimate economic value and eegional Economic effects of fishery improvements. N Am J Fish Manage. 26:301–307.

Lumb CE, Johnson TB, Cook HA, Hoyle JA. 2007. Comparison of lake whitefish (*Coregonus clupeaformis*) growth, condition, and energy density between Lakes Erie and Ontario. J Great Lakes Res. 33:314–325.

Murphy BR, Willis DW, editors. 1996. Fisheries techniques, 2nd ed. Bethesda (MD): American Fisheries Society.

Neff BD, Cargnelli LM. 2004. Relationships between condition factors, parasite load and paternity in bluegill sunfish, *Lepomis macrochirus*. Environ Biol Fish. 71:297–304.

Ney JJ. 1999. Practical use of biological statistics. In: Kohler CC, Hubert CA, editors. Inland fisheries management in North America, 2nd ed. Bethesda (MD): American Fisheries Society. p. 167–191.

Núñez D, Niklitschek M. 2010. Caracterización de la pesca recreativa en la Patagonia Chilena: Una encuesta a turistas de larga distancia en la región de Aysén. Estudios y Perspectivas en Turismo. 19:83–104.

Pangle KL, Sutton TM. 2005. Temporal changes in the relationship between condition indices and proximate composition of juvenile *Coregonus artedi*. J Fish Biol. 66:1060–1072.

Pascual MA, Macchi P, Urbansky J, Marcos F, Riva-Rossi C, Novara M, Dell' Arciprete P. 2002. Evaluating potential effects of exotic freshwater fish from incomplete species presence absence data. Biol Invasions. 4:101–113.

Pothoven SA, Nalepa TF, Schneeberger PJ, Brandt SB. 2001. Changes in diet and body condition of lake whitefish in southern Lake Michigan associated with changes in benthos. N Am J Fish Manage. 21:876–883.

Quirós R. 1998. Fish effects on trophic relationships in the pelagic zone of lakes. Hydrobiologia. 361:101–111.

Rennie MD, Verdon R. 2008. Development and evaluation of condition indices for the lake whitefish. N Am J Fish Manage. 28(4):1270–1293.

Ricker WE. 1975. Computation and interpretation of biological statistics of fish populations. B Fisheries Res Board Can. 191:1–382.

Scott RJ, Noakes DLG, Beamish FWH, Carl LM. 2003. Chinook salmon impede Atlantic salmon conversation in Lake Ontario. Ecol Freshw Fish. 12:66–73.

Shrestha RK, Seidl AF, Moraes AS. 2002. Value of recreational fishing in the Brazilian Pantanal: a travel cost analysis using count data models. Ecol Econ. 42:289–299.

Simpkins DG, Hubert WA. 1996. Proposed revision of the standard weight equation for rainbow trout. J Freshw Ecol. 11:319–326.

Steinhart GS, Likens GE, Soto D. 2002. Physiological indicators of nutrient deficiency in phytoplankton in southern Chilean lakes. Hydrobiologia. 489:21–27.

Soto D. 2002. Oligotrophic patterns in southern Chilean lakes: the relevance of nutrients and mixing depth. Rev Chil Hist Nat. 75:377–393.

Soto D, Arismendi I, González J, Sanzana J, Jara F, Jara C, Guzmán E, Lara A. 2006. Southern Chile, trout and salmon country: invasion patterns and threats for native species. Rev Chil Hist Nat. 79:97–117.

Thomasson K. 1963. Araucanian Lakes. Plankton studies in north Patagonia with notes on terrestrial vegetation. Acta Phytogeogr Suec. 47:1–141.

Trudel M, Tucker S, Morris JFT, Higgs DA, Welch DW. 2005. Indicators of energetic status in juvenile coho salmon and Chinook salmon. N Am J Fish Manage. 25:374–390.

Vigliano P, Lippolt G, Denegri A, Alonso M, Macchi P, Dye C. 2000. The human factors of the sport and recreational fishery of San Carlos de Bariloche, Río Negro, Argentina. Fish Res. 49:141–153.

Wege GW, Anderson RO. 1978. Relative weight (Wr): a new index of condition for largemouth bass. In: Novinger GD, Dillard JG, editors. New approaches to the management of small impoundments. Bethesda (MD): North Central Division American Fisheries Society Special Publication No. 5. p. 79–91.

Woelfl S. 2007. The distribution of large mixotrophic ciliates (*Stentor*) in deep North Patagonian lakes (Chile): First results. Limnologica. 37:28–36.